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Public Perception and Coastal Pollution at Identified Beaches in South Wales.

Cliff Nelson BEng (Hons), MSc

Submitted to the Open University, June 1998, for the degree of Ph.D. in Coastal Science

Author no. P9277577
Date of award: 18th September 1998

Sponsoring Establishments - University of Wales Institute Cardiff and University of Glamorgan

DECLARATION

I hereby declare that this thesis is the result of my own work and that due reference is made where necessary to the work of other researchers and authors.

I further declare that this thesis has not been accepted in substance for any former degree and is not currently submitted in candidature for any other degree.

Candidate

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Director of Studies

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Second Supervisor

A.T. Williams

Professor A.T.Williams

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Abstract

Considerable controversy exists in the world with respect to coastal quality. A multi-disciplinary project was initiated to examine the health effects of bathing in sewage contaminated coastal waters, using a popular beach resort, Whitmore Bay, close to the cities of South Wales; and to explore ways of measuring public perception of coastal pollution at selected beaches in South Wales including Whitmore Bay, Langland Bay and Cefn Sidan. The research also investigated the regulatory framework responsible for the sustainability of coastal tourism and the effectiveness of beach award flags as marketing tools in the promotion of resorts.

Current legislation addresses coastal pollution in terms of physical health criteria with little regard given to aesthetic quality of sea/landscape and psychological well-being of the beach user. It is necessary to overcome the dichotomised approach to beach management by crossing the boundaries between the physical and social sciences in order to take an holistic view of the coastal scene, accounting for environmental, political, economic and social aspects.

An epidemiological/microbiological investigation was conducted at Whitmore Bay during the summer of 1995. Statistical modelling, using Linear Logisitic Regression, indicated swimmers to significantly increase their chance of contracting an illness in comparison to non-swimmers and also identifed non-water related factors to have a confounding effect; no interaction was observed. These findings were in congruence with other major studies. Beach questionnaires were distributed to elicit information on the activities, health and socio-demographic characteristics of the subjects during the day of the survey (n=1276). A telephone interview schedule was utilised 10 days post the beach survey to investigate the differential in illness rates between the cases and controls (n=585). Water sampling was carried out on the days of the health risk survey. Although, high counts of both *E.coli* and faecal streptococci were recorded, reaching an average of 3400 and 440 per 100ml respectively, no dose response relationship was observed between morbidity rates and bacterial indicator density.

A semi-structured questionnaire was employed to obtain data on beach user perception to coastal pollution and beach award schemes for both the 1995 and 1996 surveys. The 1995 questionnaire served a dual approach running simultaneously with the epidemiological-microbiological analysis (n=1276). The 1996 survey questionnaire was developed from the original 1995 questionnaire, and distributed at an additional two beaches in South Wales, Langland Bay and Cefn Sidan, (n=821). Results of both surveys showed that beach users were acutely aware of coastal pollution both land based and marine and suggested that public awareness of the different beach award schemes is low. Of the different types of award systems included on the questionnaire, the European Blue Flag Award gained highest recognition (26-30%), but even those that identified with it often had a misunderstanding of its true meaning. If consumers misinterpret the meaning of the flag which flies on a designated beach, then the designation of the beach will do little to offset consumers' concerns about health risks.

To support the questionnaire interviews, litter surveys, formed around the Norwich Union Coastwatch study were conducted and Secchi disc readings were also taken at the three beaches to obtain data on both beach and marine aesthetic indicators. High quantities of litter were recorded, in particular plastics and polystyrene, deposited mostly by visitors. Also, higher levels of turbidity proved to negatively impact the perception of water quality and have an effect on beach user behaviour.

Results of the study highlighted the importance of understanding the cognisance of the beach user in evaluating beach and waterscapes, taking account their experience and expectations, and the vacuum which exists between decision makers and the general public. A conceptual model was designed to describe the beach management process creating a flexible management framework which encompass all key variables, their interdependency and facilitate their measurement.

The implication to management is to challenge the ineffective intellectualised approach currently in operation, identifying all stakeholders in the planning process, including the public and private sectors and the consumer.

Chapter 1 Introduction

1.1 Introduction

Sun, sea and sand have proved an attractive cocktail to tourists since Victorian times, providing an irresistible destination for both holiday makers and recreationalists. During this era a day at the seaside was perceived to be a health activity and *'trips to immerse oneself in salt water were done so with almost missionary zeal'* (Rees, 1993 p.16). A stark contrast exists today with the alleged unhealthy condition of British beaches (The Sunday Times, 1992; Wales on Sunday, 1994; Western Mail, 1995), which has been brought to the publics' attention via the media's thirst for sensational news and the political activities of high profile, informed campaigning by individuals and organisations, such as Surfers Against (SAS, 1995a). Use of emotive headlines this decade such as 'You might as well take an ice cream to the toilet' (The Times, 1994) and 'On the trail of the Mumbles Monster' (The Times, 1994) have evoked quite different attitudes to a day at the British Seaside.

Continued urbanisation of the coastal fringe (MCS, 1997a), expanding water-based recreation and development of high technology sporting equipment, coupled with increased disposable leisure expenditure is increasing pressure on the natural coastal environment (Borrego, 1996; Ballinger, 1997). Another implicit result of these developments will be greater prominence placed on health related hazards from contact with coastal waters. It is therefore of great importance that regulatory bodies recognise the necessity to reconcile anthropogenic demand with public health and commitment to sustainable management planning. The World Health Organisation (WHO) endorse this requirement through the European Charter on Environment and Health (WHO, 1989a) which illuminates the need for a clean harmonious environment for good health. Strategic guidelines laid down in the Charter outline responsibilities of governments, by highlighting the importance of aesthetic and social factors, frequently omitted from current research (Phillip, 1994a).

1.2 Coastal Pollution

Water covers 71% of the earth's surface and is a vital component for mans' existence. Couper's (1990) view is that the ocean is the last major frontier on earth for the exploration and development of resources to sustain mankind into the future. It is also argued that disregard of the importance of this natural resource will lead to irreversible and dire consequences. In particular persistent utilisation of rivers, estuaries and the sea for the disposal of waste has a wide range of serious implications for the health of the marine environment - delineated through the United Nations definition of marine pollution:

> *'The introduction by man, directly or indirectly, of substances or energy into the marine environment (including estuaries) resulting in such deleterious effects as harm to living resources, hazards to human health, hindrance to marine activities including fishing, impairment of quality for use of sea water, and reduction of amenities'.*
> (GESAMP, 1982 p.8).

This definition covers a spectrum of issues including the effect on health and reduction of amenity value of the sea. Overuse of the sea as a source for dumping is having a severe detrimental impact on the aesthetic value of our coastline and the health of beaches and those that use them. Public attitudes towards coastal pollution and the indiscriminate use of the marine environment as a rubbish tip are changing. Clark (1992) stated that environmental expectations are high and are continuing to rise in respect of waste disposal practices. A concern of this research is to view the extent to which discharge of waste and littering of beaches impair the coastline and are potentially hazardous to health.

In particular coastal bathing water quality is of universal concern. Rees (1994) acknowledged the complex problems encountered in designing effective epidemiological studies, which amongst other things, include standards and their interpretation, public perception, official explanations, scientific discord and a healthy dose of politics.

National governments and the European Union have become increasingly involved in trying to find solutions to the problem of worsening water quality. Legislation setting standards for European waters is hinged around two main Directives, the EC Directive concerning the quality of bathing water (CEC, 1997) and secondly the Directive concerning urban waste water treatment (CEC, 1991). The Bathing Water Directive (CEC, 1997) is the most prominent of the two mentioned with specific regard to bathing waters. It has been the cause for much disputation since its initial introduction over 20 years ago (CEC, 1976a). The Ammended Bathing Water Directive (CEC, 1997) now in place, came in to force force on 31st December, 1997. Compliance is based around achieving set values defined for 19 determinands. Two standards are set for most of the parameters, both a Mandatory and stricter Guideline value. The two microbiological parameters are considered to be the most appropriate determinands for indicating faecal contamination of bathing waters. They are escherichia coliforms (*E. coli*) and faecal streptococci, with Mandatory standards of 2,000 per 100 ml and 100 per 100ml respectively.

The original Bathing Water Directive (CEC, 1976a) was criticised for the lack of proven epidemiological evidence in its design criteria, in particular selection of inappropriate bacterial indicators (Kay *et al.*, 1990; Phillip, 1991). Increasing public awareness over environmental concerns has added to the impetus behind reforms to current legislation over recreational waters. The most prominent change has been the inclusion of an Mandatory standard for faecal streptococci, replacing the total coliform parameter. It is now widely accepted that faecal streptococci is a better indicator of health risk than both total coliforms and *E.coli* (Kay, 1986). In the UK the Environment Agency, an independent regulatory body, is responsible for monitoring bathing waters and ensuring compliance with EC regulation. It is likely that the new Mandatory standards for faecal streptococci will markedly increase the number of UK bathing waters failing to reach EC standards.

In addition to health aspects of beaches, the EC Bathing Water Directive also sets the standard for the formulation of many seaside award schemes. The European Blue Flag is the most noted of these and is found in all Member States (FEEE, 1997). The Tidy

Britain Group (TBG) act as the UK agent for these systems. For a beach to receive a flag they must qualify with a series of quality criteria laid down by the designer of the respective award. In combination with the Blue Flag which is aimed purely at resort beaches, the TBG also offer their own seaside award flags (TBG, 1997a) which are geared towards catering for rural beaches as well as resort beaches. The other main difference between the two systems is that the water quality requirement of the Blue flag is the guideline criteria set in the EC Bathing Water Directive, whereas the TBG awards only require attainment of the mandatory standards. From a management perspective these systems are intended to encourage the improvement of beach quality for users and as an aid to tourism destination marketing.

1.3. South Wales - Case Study

Wales has a great natural beauty with a diverse coastline of *circa* 1600km in length (MCS, 1997b), of which 70% had environmental designations (Ballinger, 1997). Williams (1996a) describes South Wales in particular as being fringed by an immensely varied coastline which ranges from high sea cliffs to low energy sedimentary embayments and estuaries. Tourism, always an important part of coastal resort economies has become an important element of the Wales economy particularly since the decline of traditional manufacturing, coal and steel industries. A large proportion of tourism in South Wales is centred around the coast, and its beaches, which also offer an ideal environment for a wide range of water sports (Nelson, 1996a, 1996b). The decline of the British two week holiday at the seaside has been offset to some extent by the growth in activity holidays which depend to a certain degree on the quality of the natural environment. Consequently, increasing awareness of environmental issues and in particular coastal pollution could have a major bearing on tourism markets and the local economies of seaside towns (Nelson, 1996b). South Wales is heavily reliant upon its beaches for tourism and provides ample evidence of the influence of the environmental agenda with respect to two recent marine pollution related incidents. First, the oil disaster early in 1996 when the *Sea Empress* tanker was grounded off the Pembrokeshire coastline. The tanker spillage (*circa* 70 ktons of crude oil), the biggest in UK waters, (Mair, 1997) spread across 30 miles of the Pembrokeshire coast having both immense environmental and social implications (Inshore, 1996). In the first instance the pollution badly affected the bird life, suffocated the benthic communities of the sea bed and consequently affected the food chain. The economic impact led to the fishing industry being forced to a standstill and an immediate reduction in tourist numbers for 1996 season. It has been estimated that the cost of the disaster in tourism terms was approximately £100m (The Daily Telegraph, 1998).

Second, two years prior to the *Sea Empress* catastrophe, two severe cases of neurological disabling symptoms experienced by two teenagers came to light, supposedly contracted from swimming in South Wales (Wales on Sunday, 1994a). The beach in question was Oxwich Bay, Gower, well known for its impeccable water quality (NRA, 1986-1994; Environment Agency, 1995-1997). No causal proof has been obtained extrapolating their disease back to originating to the water (Wright, 1995). However, speculation has increased over the health risk associated with swimming in coastal waters and the Oxwich cases have also called into question the effectiveness of the EC Bathing Water Directive (CEC, 1997) in safeguarding public health after the Bay's consistent compliance with the current Guideline standards, and attainment of the European Blue Flag. This poses the argument of whether current legislation ensures adequate protection of health, on which most of the existing beach award schemes are based, including the Blue Flag Award (FEEE, 1997) and the Tidy Britain Group (TBG) Seaside Awards (TBG, 1997).

1.4. Background to Literature Research

A substantial amount of work has been carried out in assessing coastal pollution and bathing water quality. These studies can be broken down into two main categories, marine and land-based. From a water quality perspective the current research are mostly epidemiological/microbiological based concentrating on health risk from bathing (Cabelli, 1983; Lightfoot, 1989; Pike, 1994). These studies, reviewed in Chapter 3, have attempted to quantify health risk from exposure to seawater and consequently set objective water quality standards to protect the health of bathers. The most prominent of these standards are bacteriological determinands, discussed above, designed to indicate faecal contamination of coastal waters. Shore-side surveys on beach quality have been conducted which have tried to quantify beach litter (Dixon and Hawksley, 1980; Rees and Pond, 1994; Pollard, 1996a, 1996b) and identify hazardous items to health (Dixon and Dixon, 1985; Phillip *et al.,* 1997). Managing beach litter is a very complex and international problem due to the source of debris often being marine borne (Scott, 1972; Williams and Simmons, 1995; TBG, 1997b). For example, litter found on the coast of Cumbria was extrapolated back to originating from 27 countries (TBG, 1997b). Present methods of dealing with litter pollution are curative rather than preventative, i.e. tackling the problem at source. Clearing beaches is usually the responsibility of local authorities, who frequently use mechanical rakes during the tourism season (VOG, 1996a). In certain regions voluntary schemes are organised to tidy the coastline, such as the Readers Digest Beachwatch Campaign run by the Marine Conservation Society (Pollard, 1996b).

It has been proved that perception of debris along river banks is intrinsically linked to perception of river water quality (Dinius, 1981; Smith *et al.,* 1995a). The WHO (1994a) extended this theory by postulating that poor beach aesthetics are often interpreted by the public as inferring poor chemical and microbiological quality of water. Public perception of riverine quality and fresh waters are fairly well documented (Burrows and House, 1989; House and Sangster, 1991; Smith *et al.,* 1995a), but few attempts have been made to gauge perception of the marine environment (Phillip, 1990, 1994a; Green and Birchmore, 1993; Williams and Nelson, 1997a). The dearth of literature on the marine environment extends to the aesthetic quality of coastlines and the way in which coastal pollution is perceived. The World Health Organisation have identified that along with phsyio-chemical, ecological and socio-economic aspects of environmental impact aesthetic factors are of prime importance which need to be accounted for (Phillip, 1994a). It is argued that failure to accommodate the psychological welfare of the consumer will ultimately lead to an economic loss to tourism (Phillip, 1994a).

1.5 Development of the Research Principles

1.5.1. Coastal Zone Management

Integrated coastal zone management is in its infancy within Wales, (Ballinger, 1997) compared to the USA and Canada who are at the vanguard in recognising the importance of protecting their precious coastal resources (Harvey, 1988). The *'Beach'* is a sub-ecosystem operating within a dynamic coastal environment, supporting a wide diversity of flora and fauna and a recreational destination for man. In South Wales this system is particularly dynamic and energetic due to the exceptionally high tidal range, approximately 16m (Hydrographic Office). It has been demonstrated that aspects of beach management have been challenged through the research process, but generally in a piecemeal and unintegrated fashion (Williams and Davies, in press). Coughlin (1976) noted that little attempt has been made to combine objective water quality monitoring with public perception of water quality. Some work has been carried out on rivers investigating water quality and public perception, such as that done by Moser (1984). But again a paucity of literature exists on these issues concerning the marine environment. The only study designed specifically to examine the relationship between microbiological measurements of the sea and public perception of water quality was conducted by the Robens Institute (1987). At the onset of this study it is interesting to note their explanations for the lack of previous work. They argued that it was due to the inherent difficulties with such investigations. This thesis explicitly sets out to solve some of those difficulties in its design.

It is hypothesised in this research that beach management needs to recognise the inter-relationships between objective water quality and health risk from swimming, the less tangible and subjective perception of coastal pollution in the eyes of the consumer and the interaction with actual beach behaviour. In addition comprehensive beach management is not complete unless economics and sustainability are added to the equation. The WHO (1994a) confirm the need for sustainable multidisciplinary and intersectoral efforts in particular for enhancement of water and bathing beach quality and benefits for public health. This thesis identifies the increasing need to cross the boundaries between the physical and social sciences in order to take an holistic view of the beach environment to tackle these issues. Drive to undertake this research was derived from the WHO (1989a) in The European Charter on Environment and Health, which underpins the above philosophy, recognising the requirement of a multi-disciplinary approach designing studies involving health risk analysis and environmental issues by stating:

> *'Interdisciplinary research programmes in epidemiology with the aim of clarifying links between the environment and health should be encouraged and strengthened at regional, national and international levels'.*
> (WHO, 1989a p.4).

1.5.2. Seaside Award Schemes

This research further investigates the effect of beach classification with seaside award schemes such as the European Blue Flag (FEEE, 1997) and the Tidy Britain Group Seaside Awards (TBG, 1997), discussed above, which have become pivotal in the marketing of coastal resorts. These schemes aim to monitor water quality and give objective advice and information on issues considered by their sponsors to be central to public safety, peace of mind and enjoyment. However, it is postulated that their profusion and complexity may produce entirely the opposite effect. The assumption is that recognition of a beach in awarding flag status influences the consumer, but there is no evidence to support the influence on beach choice or beach-user behaviour. Research has suggested that perception is of considerable influence in tourist behaviour (Botterill *et al.*, 1991). It is hypothesised that the 'Perception of Pollution' is of more importance in influencing tourist behaviour than 'flag' designation. Few studies have addressed the issue of seaside award schemes (Williams and Morgan, 1995; Nelson, 1996; Nelson *et al.*, 1997; Owen *et al.*, 1997; Nelson *et al.*, in press (b)) and their effectiveness as beach marketing tools. Morgan (1996) noted the intellectualised 'top down' approach taken in designing these systems with little or no regard for the end user. In line with his work this study tackles the problem of evaluating the effectiveness of flag schemes from a 'bottom up' approach, in an attempt to establish from the consumers themselves their needs for and understanding of the present schemes.

It is argued that perception of coastal pollution goes beyond the confines of what is and what is not a polluted beach. Public fear about bathing in sewage contaminated seawater could have a significant detrimental impact on the Welsh tourism economy. A coordinated strategy is required to measure the effect of public perception to coastal pollution and the subsequent effect on tourism. The WHO (1989a) recognised the responsibility of regulatory authorities over management of natural resources, by stating the need to establish links between environment and health at all levels of government and regulation, filtering through supranational levels such as the EU to national governments and finally at regional level. South Wales has been used as a test case to found the study which, as mentioned above has been at the forefront of environmental issues. The research is coincident with recent developments in Wales through the Green Seas Initiative set up by the Wales Tourist Board (WTB), to improve bathing water quality and jointly funded by Welsh Water (Dwr Cymru, 1997). Approximately 30 other organisations are involved in the project which aims to clean up the water quality around the Welsh Coast through improved sewerage systems. The self-proclaimed criteria for the measurement of success of the Green Sea Initiative is the attainment of 50 European Blue Flags at Welsh beaches by the Millenium. The importance of this research is therefore accentuated as the levels of understanding and conusmer perceptions of the flag schemes themselves will test the salience of the performance criteria set for major public/private sector investment programmes. The results of the research have, therefore, an important public policy dimension.

1.6 Aim and Objectives

The aim of the research is to:

To examine the complex relationships relating coastal pollution to: public perception; susceptibility of beach users to illness; behavioural patterns and attitudes to seaside award schemes and the regulatory framework through which they interact.

To achieve this aim the following seven objectives have been outlined.

1. Examine the health risk from bathing in Whitmore Bay
2. Analyse the water quality at Whitmore Bay
3. Investigate if a dose response relationship exists between illness rates from exposure to seawater and faecal indicators at Whitmore Bay
4. Investigate beach user perception to seawater pollution and investigate potential correlations with objective water quality at 3 identified beaches in South Wales
5. Investigate beach user perception to beach pollution and investigate potential correlations with objective water quality at 3 identified beaches in South Wales
6. Identify aesthetic indicators of coastal pollution
7. Assess the effectiveness of beach seaside award schemes in marketing of coastal resorts at the three beaches
8. Investigate if a relationship between beach behaviour and turbidity exists across the three beaches

1.7 Methodology

This research is both multi-disciplinary in that it draws upon quite distinct bodies of knowledge and their respective empirical methods and inter-disciplinary in that it attempts to explore the dynamic inter-relationships of physcial and social variables encountered when studying the beach environment. Figure 1 maps the mechanism to achieve the objectives set in this study. Chapter 6 gives a comprehensive details of the methods selected.

1.8 Research Schedule

The water quality field work covering objectives 1-3 was carried out at Whitmore Bay, Barry, during the summer months of 1995 and further water quality sampling was conducted in September 1996. Initial investigations using a semi-structured interview into the public perception of coastal pollution were also carried at Whitmore Bay during 1995. The interview schedule was developed to include another 2 identified beaches in preference to further epidemiological/microbiological work. They were Langland Bay (Gower Peninsula) and Cefn Sidan (Pembrey Country Park). The summer months of 1996 were utilised to continue the field work covering the 3 beaches, again using a semi-structured questionnaire. In addition visual props were used as stimulii to gauge perception of litter items and awareness of seaside award schemes.

The timing of this project has been fortuitous in light of the media attention and increasing public awareness of pollution on British beaches in conjunction with the dynamic legislative process of reforming the EC Bathing Water Directive. In more localised terms there has been major outcry over two particular incidents in the Welsh Principality and also exciting developments such as the Green Sea Initiative and Coastal Forum in Wales.

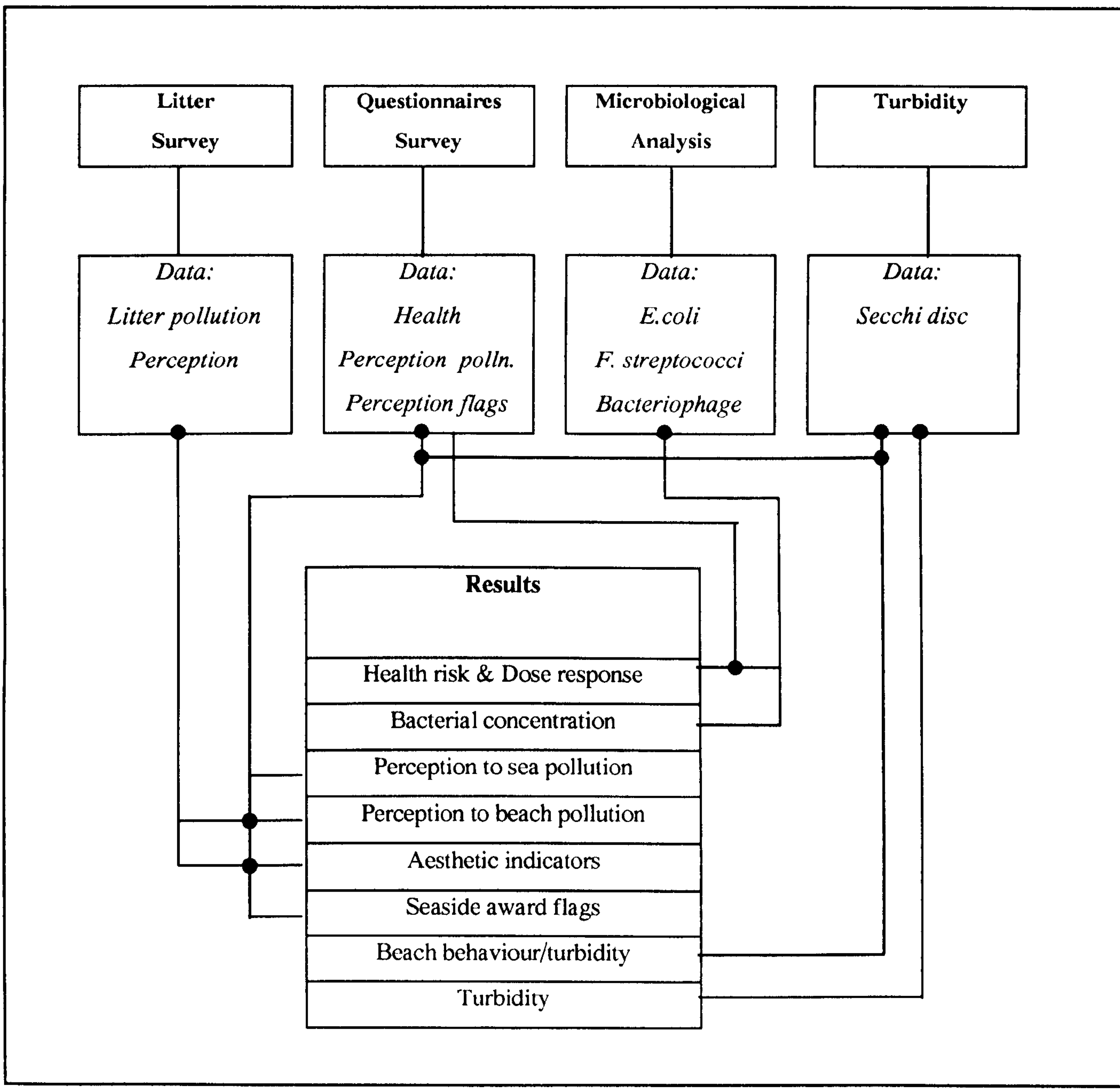

Figure 1 Diagrammatical Representation of the Methodology

Chapter 2 Physical Background

2.1 Beach Selection - South Wales Coastline

A cross-sectional methodological approach to the study design was employed and a set of criteria selected to define the type of beaches required for the survey work. South Wales beaches were stratified using information from the Tidy Britain Group (TBG) to display both the European Blue Flag (FEEE, 1994) and Tidy Britain Seaside Award flag status and non flag status (TBG, 1994a). Consideration and classification of unidentified beaches into rural and resort proved there were none that fall into the resort category along the SW coast, and those that fit the rural category tended to be small and ill frequented, even during summer. This would inevitably cause problems obtaining a significant number of participants during questionnaire analysis. The stratification process also chose beaches which had a graduation of water quality so that a relationship between pollution indicators and relative health risk could be ascertained. Other important attributes of the selected beaches were popular, well defined and compact to assist interviewing, attracting a mixture of visitors and residents, affected if at all by a single point source of pollution rather than by storm sewerage outflow and close proximity to laboratory facilities.

Three locations along the South Wales coastline were chosen to carry out field work, Whitmore Bay, Langland Bay and Cefn Sidan. An intensive water sampling strategy was designed and carried out at Whitmore Bay, which was in close proximity to the microbiological laboratory as opposed to using a limited sampling programme analysing all three beaches. The beaches lie on the Bristol Channel, a quasi-estuarine zone which experiences approximately 2 tidal cycles every 24 hours. This is the largest range in Europe and second largest in the world, next to the Bay of Fundy in Canada (Severn Estuary Strategy, 1997). Between Cardiff and Avonmouth the tide reaches its maximum range of 16.4m with a measured 27 knot tidal race at Nash Point (Hydrographic Office). All three beaches are actively being eroded experiencing a net loss in the sediment budget (Williams *pers.comm.*, 1997a).

2.2 Whitmore Bay, Barry Island

Barry Island is situated on the southern tip of Wales, in the Vale of Glamorgan (VOG), lying 10 miles west along the coast from Cardiff the capital city (Nelson and Williams, 1997). Research was conducted at Whitmore Bay the larger of two designated beaches situated in the port town of Barry. The beach is a pocket bay formed through the erosion of Carboniferous Limestone between two headlands composed of the grey limestone. The shore is predominantly sandy, grain size 2.5ϕ (0.25mm) (Williams *pers.comm.*, 1997a), south facing with a large surface area of 200,000m^2, 800m long and 250m wide to low water; OS reference ST: 115 663. Whitmore Bay is backed by a Victorian promenade separating the beach from a highly developed hinterland. Barry Island is a popular destination for holiday makers, day trippers and locals, providing a residential holiday camp, funfair, shops, amusements and numerous bed and breakfasts. In addition to the holiday camp there consists a funfair, shops, pubs and amusements. Tourism is very important to the area, the beach attracting 850,000 people during 1994, providing 13.4% of the employment sector (VOG, 1996a). The surrounding town of Barry with a population of approximately 46,000 makes up 41% of the Vale of Glamorgan Borough Council providing a large catchment. Close proximity to the M4 also provides ease of access for day trippers, a large number coming from the South Wales Valleys and Gwent region.

The seawater is very turbid often containing visible floating items and sewage related debris. There are two sewage discharges to Barry waters, East and West, serving populations of 34,600 and 26,000 respectively, both currently only receive a screening process. Welsh Water have pledged that all beaches in Barry will comply with the EC Bathing Water Directive (CEC, 1976a) by 1998, after completion of £150m sewage scheme in Cardiff and the Barry Bathing Water Scheme which will see the installation of a new system of treatment for the surrounding area (Welsh Water, 1997). In the past Whitmore Bay has struggled to comply with European bathing water standards. Since 1986 to date the bathing waters have only had a 46% pass rate, meaning it also fails to meet requirements for a Blue Flag or TBG Seaside Award (Environment Agency, 1997). Management of Barry Island and Whitmore Bay is undertaken by the VOG Borough

Council. Currently no coastal management plan exists (VOG *pers.comm.,* 1996b), but provision of lifeguards and daily beach cleansing operations are provided by the Council. Plates 2.1 and 2.2 were taken of the beach during hot weather in August 1996. Figure 2.1 shows Whitmore Bay and surrounding Barry displaying water sampling points (S1 and S2), litter grid positions (G) and transects (T1,T2 and T3). Also shown are low water mark and high water mark, where the litter trawl investigation was done. Offshore sewage discharges are also represented.

Plate 2.1- West View of Whitmore Bay

Plate 2.2- East View of Whitmore Bay

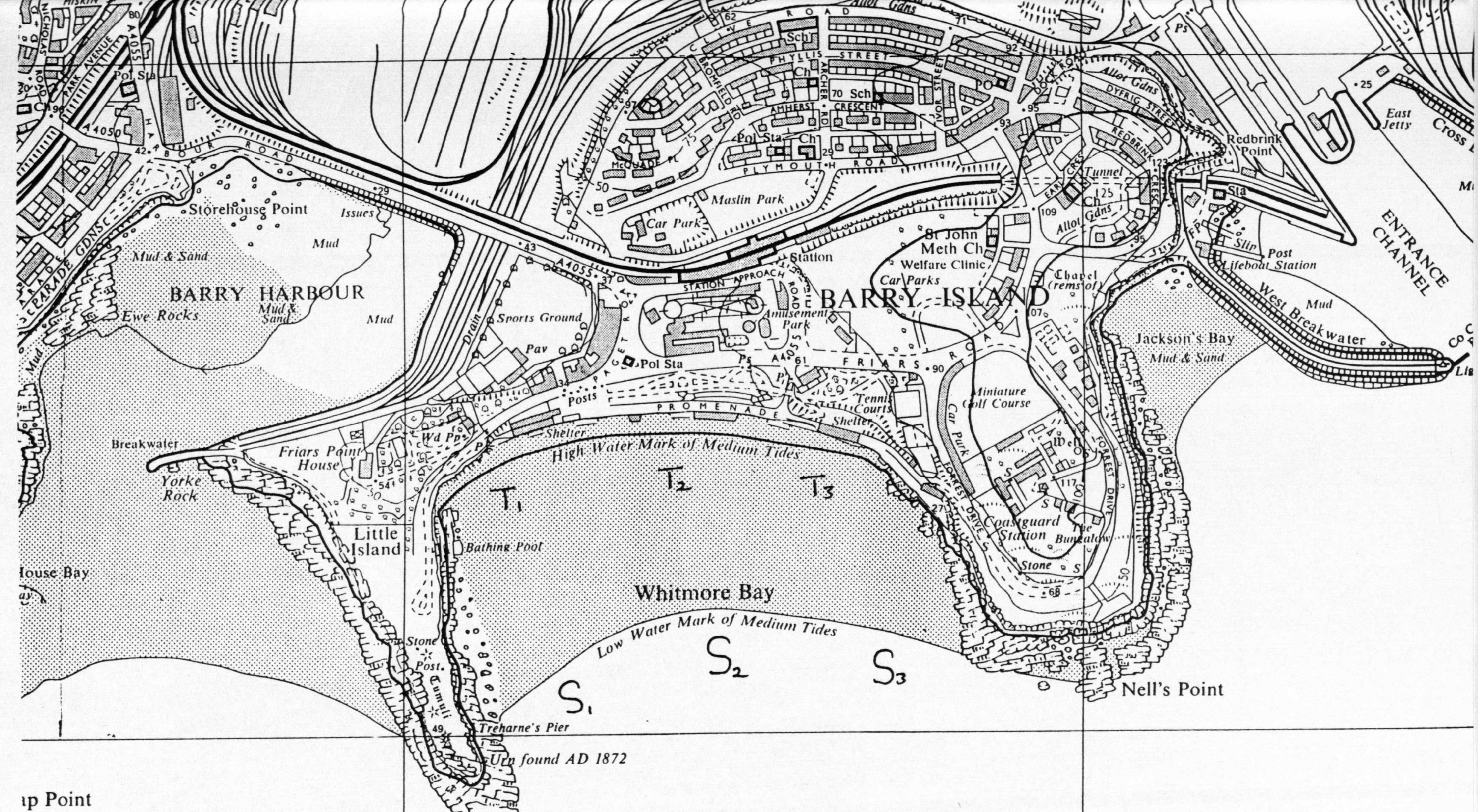

Figure 2.1 - Whitmore Bay With Sampling Sites

2.3 Langland, Gower

Gower is a peninsula to the west of Swansea, within the boundaries of City and County of Swansea Council. Gower has a very attractive landscape and large diversity of wildlife which has been recognised nationally, becoming Britain's first designated Area of Outstanding Natural Beauty (AONB) in 1956 (Morgan and Williams, 1995a). There is a wide variety of scenery on Gower, which is particularly unusual for a such a small area. The coastline is of significant importance, 54 km of which has been designated as Heritage Coast. The area contains a wide spectrum of conservation interests and landscape features, including limestone cliffs, sandy beaches, sand dune systems and salt marshes (Nelson, 1994). Its flora is more diverse than any other area of comparable size in Britain (SCC, 1990). There are three National Nature Reserves (NNR) and nineteen Sites of Special Scientific Interest (SSSIs) on Gower in addition to various Local Nature Reserves (LNR) and nineteen Nature Reserves set up by the Glamorgan Wildlife Trust and one by the City Council (SCC, 1990).

Tourism plays a substantial role in the rural economy of Gower with a large proportion of caravan sites and guesthouses, but is highly seasonal. Its close proximity to the M4 motorway makes it accessible to a large population, and within 4 hours travelling time of 18 million people (SCC, 1990), which accounts for the large number of visitors and tourists it receives each year. The outstanding beauty of its coastline and varied environmental conditions make it very popular for coastal recreation, ranging from surfing, water skiing and cliff climbing to swimming and sun bathing (Nelson, 1994).

The southern and western coastal areas are under significant pressure from the growing number of visitors and tourists, and have virtually reached accommodating capacity, if not exceeded in some regions (Nelson, 1994). Without appropriate management there is a risk of the 'self destruct theory', in which sheer volumes of visitors could possibly destroy the natural qualities that they wished to see and enjoy (Nelson, 1994).

SCC in consultation with West Glamorgan County Council (now the City and County of Swansea Council), the Countryside Commission, Nature Conservancy Council and the

National Trust drew up the Gower Management Plan (GMP) which was published in 1990 (SCC, 1990). The purpose of the GMP (p.1) was to supplement the Swansea local plan with:

> *'the aims of minimising the potential conflicts and the improvement of coordination of conservation effort within the context of overall goals and aims for local planning on Gower'.*
> (SCC, 1990 p.1).

The GMP is mostly land based and has been used as a starting point for effective management on Gower. However, it does not address the offshore perspective or forward any objectives for management of the coastal zone, except in the management of tourism.

2.3.1 **Langland Bay**

Langland Bay is a fine grained sandy beach, grain size 2.5ϕ (Williams *pers.comm.* (a)), which lies in the most intensively used stretch of Gower coastline (OS grid reference SS: 603 867). Plates 2.3 and 2.4 display Langland Bay during hot summer weather in August 1996. Figure 2.2 shows the Bay's location in Gower and includes points selected for the turbidity readings (S1, S2, S3). Langland is protected on both sides by Carboniferous Limestone headlands (Morgan, 1996), the western side siting a golf course, and is backed by beach chalets, tennis courts and a car park. The Carboniferous Limestone rock formation has been eroded making the beach a pocket bay. The surrounding area includes Bishop's Wood and contains a Local Nature Reserve and 2 Sites of Special Scientific Interest (SCC, 1990). West Glamorgan County Council designated the Bay as an intensive zone aimed at protecting and enhancing the coastline whilst providing facilities to cater for visitors. The Bay and surround provides a café and small gift shop selling food and drinks. Closeness to the built up area of Swansea and being within 30 minutes drive from the M4 makes it an ideal destination for day trippers. An estimated 2 million visitors are attracted to Gower each year (Mullard *et al.*, 1996). Langland is also well known for its excellent surf and ideal conditions for water sports (Nelson, 1994).

Water based activities which take place include surfing, bathing, canoeing, windsurfing and recreational fishing.

Due to the attractiveness of Gower and the beaches, it becomes subject to intense visitor pressure during the summer months (Nelson, 1994). Southern Gower from Mumbles to Port Eynon, covering Langland is the most extensively used for tourism. To control visitor numbers the City and County of Swansea Council have no plans to improve the road network onto the peninsula allowing current access to be self regulating. The water quality at Langland varies from year to year. Since 1986 to 1997 nine passes and three fails (NRA, 1995, 1996; Environment Agency, 1997) have occurred against the Imperative standards set by the EC Bathing Water Directive (CEC, 1976). Sewage discharges close to the Bay occur at Mumbles (140,747) which receives screening and Bishopston (2,000) subject to secondary biological treatment (Welsh Water, 1995). Figures in brackets represent populations served. Construction of an £80m full treatment plant in Swansea, to be fully implemented by 1997, will include UV treatment and be discharged through a 3.5 km pipe (Welsh Water, 1997), which should improve the water quality along the southern beaches. Due to the variability of Langland's water quality it did not receive either an EC Blue Flag or TBG Seaside Award Flag during the periods of this study, 1995-1997 (TBG, 1997a; FEEE, 1997). Cleansing of the beach is conducted by the City and County of Swansea Council using a motorised rake each morning and professional lifeguards are employed through the summer months along with a voluntary surf lifesaving club (Cunningham *pers.comm.,* 1994).

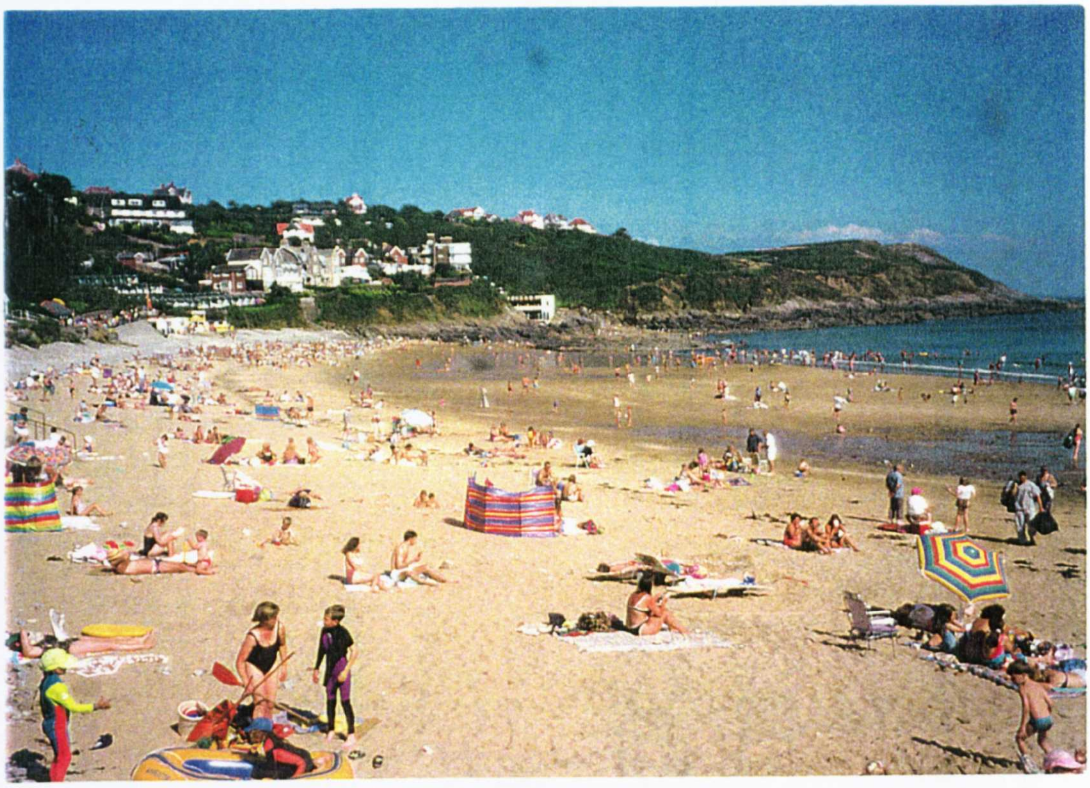

Plate 2.3 - East View of Langland Bay

Plate 2.4 - West View of Langland Bay

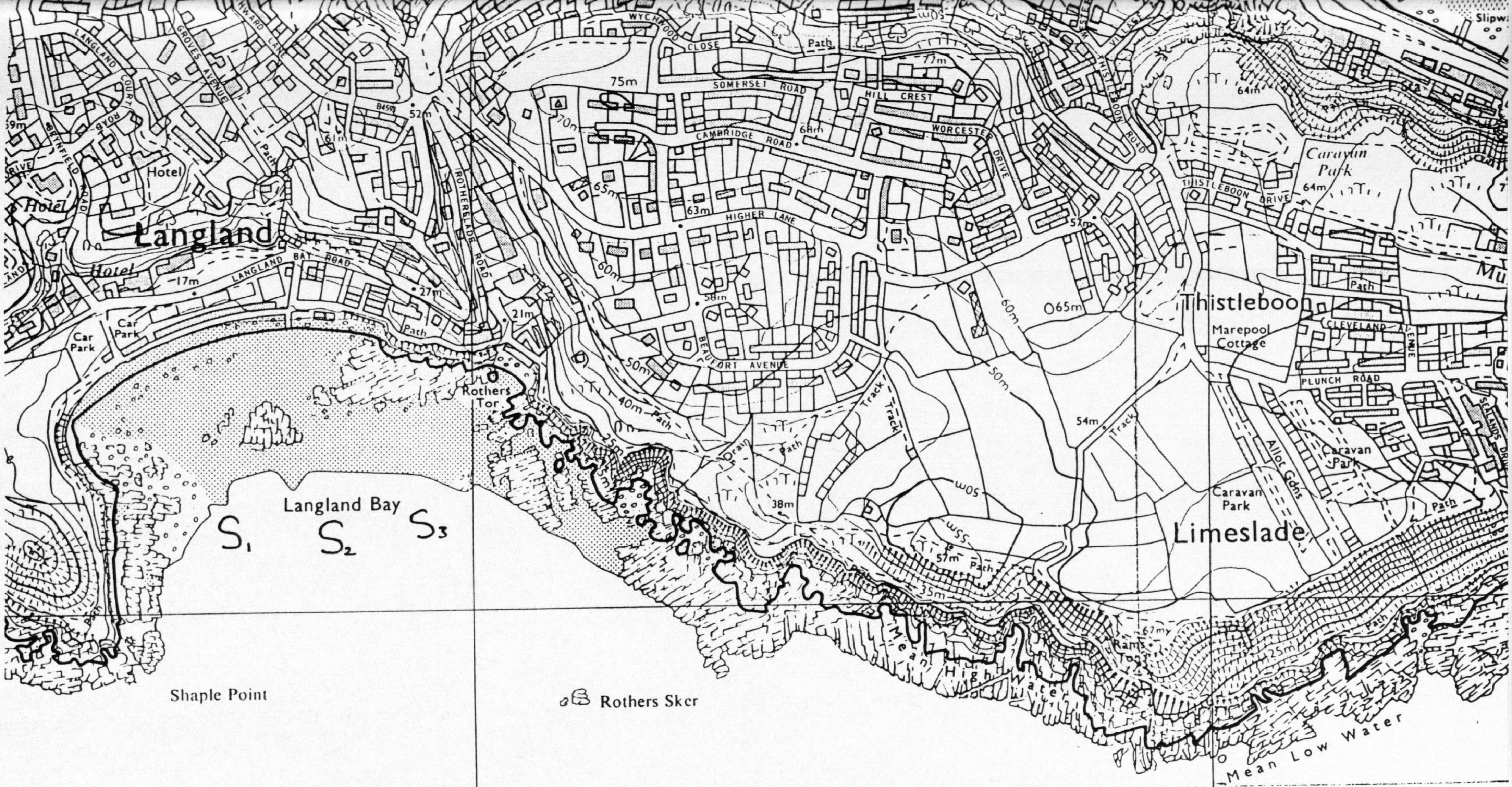

Figure 2.2 - Langland Bay With Sampling Sites

2.4 Cefn Sidan, Pembrey Peninsula

Cefn Sidan, translated as Silken Ridge in English is a beach situated in Pembrey Country Park (OS grid reference SN: 317 072). Plates 2.5 and 2.6 show the beach at Cefn Sidan. Figure 2.3 shows the beach in the context of the Peninsula displaying the points used for the turbidity readings. The Park on the Pembrey Peninsula covers an area *circa* 520 acres (Priest, 1986a) with a diversity of landscape including saltmarsh, dune system and foreshore (Nelson and Williams, 1997). The beach is the main attraction but camping facilities are available and the woodland is also enjoyed by recreationalists. Although visitors to the Park are highest during summer months all year round facilities are available including a visitor centre, narrow railway gauge, boating lake, 'pitch & putt' and an artificial ski slope (Priest, 1986b). Since development of facilities reconciled with the natural beauty of the Park an annual flux of 350,000 visitors can be expected, 945,000 were received in 1996 (Pembrey Country Park Administration Records, 1996).

Cefn Sidan beach forms the seaward edge of the Pembrey Peninsula created by marine and alluvial deposits (Priest, 1986a). The fine grain sandy beach, grain size 3ϕ (0.125mm), is large extending 10km in length from the Pier at Burry Port in the south east to the Gwendraeth River in the north west, backed by a dune system. Due to the large tidal range mean difference between high and low water is almost 1.6km, with a height of 8.7m (Hughes, 1986). To the west from the Loughor estuary is an extensive dune system, some ridges are over 30m in height. The River Towy provides sand and coastal erosion from the eastern front is carried by south westerly winds to form the dunes (Priest, 1996a). To the north east is a wide expanse of alluvium sediment forming the Kidwelly and Pembrey flats. The geomorphology of the Peninsular is still in a very dynamic phase with the coastline continuously changing. Dunes are still being formed to the south west and to the north extensive salt marshes are still developing. A sand bank is also spreading eastwards creating and 'embryonic' salt marsh known as the Pembrey Saltings (Priest, 1986a).

The urbanised communities of South Wales provide a main source of visitors as well as locals from Pembrey village and the surrounding area near Llanelli. Camping facilities allow for holiday makers which can reach the Park from the M4. Management of the Peninsula, Country Park and beach is undertaken by Carmarthenshire County Council, formerly Llanelli Borough Council. Rangers are employed all year with administration staff at the visitor centre and lifeguards during summer months (Perry *pers.comm.,* 1996). The water quality at the beach frequently reaches EC guideline standards which has led to the achievement of European Blue Flag this year, which is the 9th year it has been received in addition to TBG Seaside Award Status (FEEE, 1997; TBG, 1997; Webber *pers.comm.,* 1997). In a nationwide study on the best and worst beaches in the UK, conducted by the TBG (TBG, 1994b), Cefn Sidan was ranked in the top six. Plate 2.5 displays the popularity of the beach and the flags. The nearest outfalls to the beach are at Rhossilli serving a population of 400 receiving primary treatment and to the west are Saundersfoot with primary treatment and Tenby with secondary biological treatment serving populations of 5200 and 5000 respectively (Welsh Water, 1995). A £10m UV disinfection system is currently being constructed at Tenby, which will also receive effluent from Saundersfoot (Welsh Water, 1997).

Plate 2.5 - East View of Cefn Sidan

Plate 2.6 - West View of Cefn Sidan

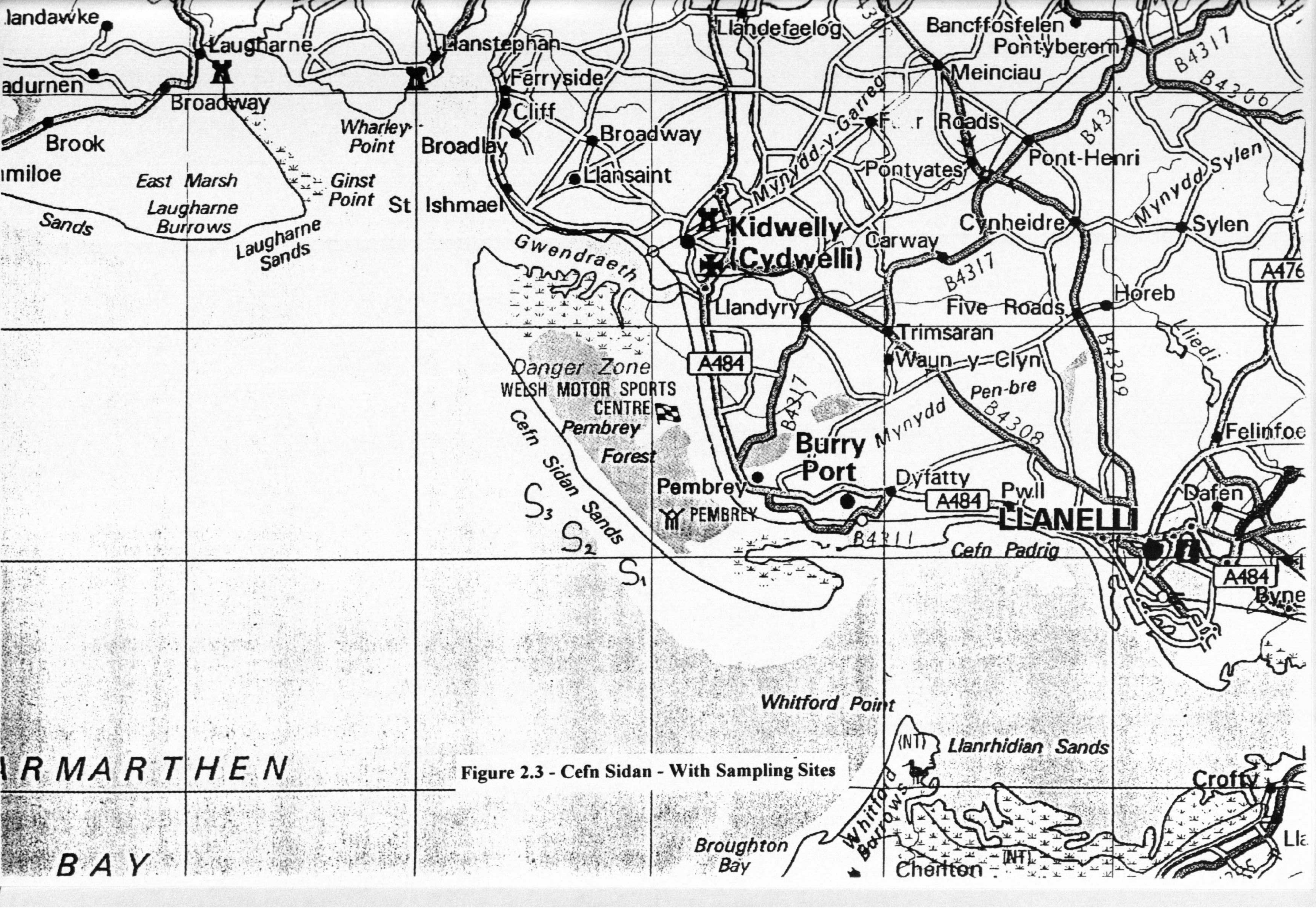

Figure 2.3 - Cefn Sidan - With Sampling Sites

Chapter 3 Water Quality

3.1 Introduction

Coastal pollution and health effects from bathing in sewage contaminated sea water are important issues (Cabelli *et al.,* 1982; WHO, 1989b; Pike, 1994). Research in this field has been limited and there is continuing debate over setting appropriate water quality standards, addressed later in more detail (refer section 3.8). Increasing media attention focused on the state of Europe's coastline, in particular the UK, is adding weight to growing public concern over pollution of beaches (The Times, 1991). International attention given to quality of coastal waters has resulted in formulation of directives and protocols, both regulatory and non-regulatory agencies to guard the marine environment (WHO, 1977, 1990a; CEC, 1997). Examples pertaining to protection of coastal waters include: the European directives on 'pollution caused by certain dangerous substances discharged into the aquatic environment of the Community' (CEC, 1976b), 'the quality required for shellfish waters' (CEC, 1979), and from a wider perspective the United Nations Environment Programme (UNEP) and WHO have introduced guidelines for protecting the ocean from pollution, for example 'assessment of the state of pollution of the Mediterranean Sea by pathogenic organisms' (WHO, 1990b).

The European Union and National governments have become increasingly involved in trying to find solutions to the problem of worsening water quality (CEC, 1991; CEC 1997). It is the intention of this study to focus on the quality of bathing waters in the context of beach pollution, which over the last decade has become an emotive and universally contentious issue (Owen *et al.,* 1997). Recent studies proving significant health risk from bathing in seawater (Lightfoot, 1989; Alexander and Heaven, 1990; Jones *et al.,* 1993) prompted reform to the current European directive concerning the quality of bathing water (CEC, 1976). Attempts to further safeguard public health are being conducted by the EC in re-setting parameters and the current framework through which water quality is controlled (CEC, 1997). A great deal of discord exists in the UK over these more stringent controls to clean up bathing beaches (ENDS 1994i). The

Government and water companies have already committed substantial investment to sewage treatment works (ENDS 1994a) to attain water quality standards laid out in the European Bathing Water Directive. The proposals have been in a state of flux for the last three years (Kay *pers.comm.*, 1997), but have now been confirmed. Change to the bacterial standards will surely increase the number of British beaches failing to comply. Implementation of this new Directive, coupled with the effect of the Directive concerning urban waste water treatment (CEC, 1991) will inevitably increase the cost of sewage treatment processes for the water service industries.

On a regional scale Wales has had to face two serious coastal issues. In 1994 bathers at Oxwich Bay (near Swansea) contracted extreme neurological problems following their visit to the beach. Two teenagers in particular were thought to have contracted their illnesses through ingestion of a virus believed to have originated from the seawater at Oxwich (The Times, 1994; Wales on Sunday 1994a). More recently the major oil disaster in Pembrokeshire added momentum to public distress over pollution on Welsh beaches when the *Sea Empress* tanker collided with rocks during rough weather on 15 February 1995, spilling 72 ktons of oil (Western Mail 1995; Mair, 1997). The economy in Wales is heavily reliant upon coastal tourism, and in reaction to these incidents the Wales Tourist Board (WTB) have instituted a project with Welsh Water called the Green Seas Initiative to counter the down turn in visitor numbers (Welsh Water, 1996). It is imperative that the effects of coastal pollution be accurately identified and an understanding of a whole range of interdependent variables, including beach aesthetics, health of bathers and tourism economics gained. It is also important to evaluate these variables in light of cost benefit analysis regarding technology to achieve a safe and healthy coastal environment, for example appropriate design of sewage treatment works to achieve European standards. However, this is a difficult proposition; even designing effective epidemiological studies focusing on health involves intrinsically complex problems, acknowledged by Rees (1994).

3.2 Sewage

Untreated or poorly treated sewage causes the majority of health problems with recreational waters and is the most common source of minor illness from water activities (Cabelli, 1983; Pike, 1994; Kay *et al.,* 1994). Disposal of waste is a continuous problem faced by man in particular domestic sewage. Originally land was used as the prime source for dumping of sewage waste. Development along the coastal fringe led to the disposal of domestic waste into marine waters (Couper, 1990) which was believed to be an economic and acceptable option. The Royal Commission on Sewage Disposal (1904) expressed that no serious injury to public health was to be expected from swimming in polluted seawater, provided reasonable care was taken in choosing the sites of sewer outfalls. In certain cases sea outfalls were only laid to high water (Clark, 1992), creating unhealthy sea conditions and substantial visual impairment of beaches.

Recent research has proved that water courses, e.g. a river and sewerage systems, are carriers of pollutants and without proper treatment they are carried out to sea (Natale, 1996). Melnick (1984) highlighted the abundant quantities of pathogenic viruses excreted into the sewerage system via faeces and urine. Prominent environmental groups, such as Surfers Against Sewage (SAS, 1995a), are driving to encourage water companies to improve treatment of domestic waste, and are gaining public support through their efforts. It is apparent that measures are required to protect both the marine environment and water recreationalists from exposure to pollution. It may be impractical to totally cease using the sea as a sink for sewage disposal, which would inevitably compound problems on land. However, careful consideration should be given to waste treatment processes. Grantham (1992) stated that to create an effective system it is necessary to first look at the aims of sewage treatment, which can be divided in to 3 main categories:

1. reduce levels of indicator organisms
2. reduce levels of pathogenic organisms
3. not result in adverse environmental side effects

The UK has to deal with huge volumes of sewage, which has led to 300 million gallons being discharged into British waters untreated daily (Rees, 1993). Most coastal outfalls are in close proximity to bathing waters, adversely effecting their quality. At present 90% of sewage entering the sea is either raw or only received maceration (Croall, 1995). Further, only 13 % of outfalls receive primary or secondary treatment (HMSO, 1990b). Various levels of sewage treatment are available, which fall into four main categories, outlined below.

3.2.1 Preliminary and Primary Treatment

Preliminary treatment involves screening the effluent to remove large objects. The primary stage is a separate process which comminutes or macerates the composition to form a slurry, which enters settlement tanks. The supernatant liquid either gets discharged to receiving waters, retaining a high biological oxygen demand (BOD) and a small percentage of suspended solids or moves on to further processes. The resultant sludge is dealt with separately. Advanced sewage treatment works decompose the sludge anaerobically forming methane gas which can be used to power the plant and the remaining material has potential to be dried to granular form and deployed as fertiliser.

3.2.2 Secondary Treatment

If future BOD reduction is required for sensitive receiving waters the supernatant liquid undergoes secondary treatment which involves filtering through a bed of rocks or coke providing a large surface area for bacteria to breakdown the organic matter. The mixture is then aerated via one of a series of different systems on the market such as the activated sludge process, consequently lowering the BOD (Clark 1992). The remaining sludge is often used as fertiliser. Although this can cause an objectionable smell, around half that is produced in Britain is used in this way on agricultural land.

3.2.3 Tertiary Treatment

If particularly high quality effluent is required, the supernatant liquid may be retained in sedimentation ponds or a grain filter system to remove further suspended solids, and then passed through ponds retaining algae to remove nitrates and/or subjected to electrolytic methods to remove phosphates.

The National Rivers Authority (NRA), whose responsibilities have been subsequently transferred to the recently formed Environment Agency in 1995, and water companies have opted for primary treatment followed by discharge of effluent via long sea outfalls. This method utilises areas of high natural dispersion, making use of the dynamic nature of coastal environments, reliant upon the motion of the tide and waves to dilute the sewage and bacterial action breaks down the organic matter. The salt water and UV light from the sun rays then act as disinfectants. The Governmental view expressed to the Royal Commission on Environmental Pollution in 1984 was:

> *'the disposal of crude sewage to the sea through well designed outfalls was not only acceptable, but environmentally preferable to alternative methods of disposal'.*
> (HMSO, 1984 p.15)

However, Grant and Jickells (1995) stated that there was an increase in densities of sewage derived bacteria at beaches near long sea outfalls. Due to the resistance of viruses to environmental stress, with survival times in seawater between 3 weeks to several months (Croall, 1995), Berg and Metcalf (1978) reported that absence of bacteria does not necessary imply absence of infectious viruses. In view of recent research the Government has changed opinion on this issue. The House of Commons Select Committee on the Environment stated that:

'the use of long sea outfalls has never been accepted enthusiastically by the public and neither has it received universal acclaim by the world scientific community. Doubts have persisted about reliance on the sea to purify and to render harmless sewage discharges, and there has been concern about the potential long-term build-up of pollutants in the sea'.
(HMSO, 1990 p.34)

3.2.4 Disinfection

There are different systems for disinfection of sewage, each with respective pros and cons (ENDS, 1992c). Chlorination is one system on the market, but can cause other problems due to production of chorine by-products. Water companies are turning to ultra-violet (UV) disinfection in preference to long sea outfalls, which appears to be an economical and effective method of complying with EC standards for bathing waters. UV treatment has been endorsed by the Environment Agency (ENDS, 1992c). Initially there were operational concerns expressed over the ability of this system to penetrate through turbid supernatant liquids. However, it has been shown that UV systems can achieve a coliform count less than 200 per 100ml in treated effluent, which is well within EC guidelines, provided the sewage has a transmissibility of 65% at a wavelength of 254nm (ENDS, 1992c). UK waters tend to be turbid due to relatively high sediment loads, rarely meeting this level of light penetration. However, satisfactory results have been achieved in Jersey with transmissibility's of 43-57% (ENDS, 1992c). Further concerns have been expressed over the microbiocidal performance of UV treatment in the reduction of pathogenic viral particles as well as the proven ability to reduce indicator organisms. Extensive trials have been in progress and the Jersey plant reports to work efficiently (personal visit, 1993). Welsh Water have now committed themselves to installing UV disinfection on all coastal outlets (Welsh Water, 1996a & 1996b). The aim is to achieve the cleanest beaches in Europe by the year 2000-2005, which entails 50 waste water clean-up projects for the entire coastline of Wales over the next five years (Western Mail, 1995) as part of a £550 million scheme (South Wales Echo, 1995). Although Welsh Water have taken the lead in the UK, a report by ENDS (1994c) states that the cost and complexity of designing new sewerage treatment works must be fully understood (ENDS 1994c). Clark (1992) endorses the high expense of implementing a modern plant with secondary treatment. Figures provided by Clark (1992) indicated the cost of a sewage processing plant for a population of 12 500, incorporating secondary treatment would be over £4 million. Up to date figures allowing for inflation would obviously increase this figure.

Although preliminary and primary treatments create aesthetic improvements, they have little effect on reducing the microbial load of sewage or reducing the number of viruses present. Raw sewage contains around 100 pathogenic viruses (Havelaar, 1993). Primary treatment coupled with a long sea outfall only transfers the microbial content to sea. Application of disinfection at source, eliminating pathogens before discharge using a short outfall pipe is a far more appropriate than using preliminary and primary treatment in combination with long sea outfalls.

3.3 Health Risk

A visit to the coast, believed to be a healthy activity both physically and mentally may have inherent health risks attached. Potential hazards include sunburn, drowning, injury from discarded litter (refer Section 4.1), and possible infection from contaminated sea water. This section concentrates on the association between water and disease which has widely been documented (Coughlin, 1976; Cartwright, 1992; WHO, 1994a). Fewtrell and Jones (1992) acknowledged that the relationship between water and disease has been apparent since ancient times. However, the issue of health from recreational use of coastal waters continues to be controversial (Kay and Wyer, 1992), with scientific opinion frequently being at variance with political points of view.

The marine environment is utilised for a wide range of diverse uses from providing a source of enjoyment for a wide range of recreational activities such as swimming, canoeing and diving to creating a sink for disposal of contaminated sewage effluent (Argardy, 1993; Steward, 1993). Such activities often conflict with each other (Harvey, 1988; Ballinger, 1997). The sea has an anthropogenic input of pathogens originating mainly from agricultural run-off and sewage disposal, which even when treated can contain a large quantities of bacteria and viruses (HMSO, 1990b), in addition to the naturally occurring microbial content. Unequivocal evidence of risk to health from bathing in sewage contaminated water exists (Cabelli *et al.,* 1982; Phillip *et al.,* 1985; Brown *et al.,* 1987; Lightfoot, 1989; Pike, 1994; Kay *et al.,* 1994). The WHO (1979) noted that recreational activities in polluted waters may cause a health hazard, including wading and boating; however, the risk of illness markedly increases when bathing and immersing ones head. This is in agreement with findings by Lightfoot (1989). The main infection route of pathogens into the body is through ingestion of water. Swimmers (on average) will swallow between 10-15ml of water each time they bathe (Rees, 1993). Exposure of mucous membranes and breaks in the protective skin barrier also act as portals for entry of pathogens (WHO, 1979).

Higher incidence of disease in bathers compared to non-bathers is not just attributable to exposure to pathogens in seawater, but also linked to the susceptibility of the host. Grantham (1992) stated that health risk due to bathing in polluted waters is reliant upon three main factors:

1. the health of the community served by the local discharge(s)

2. the bathers resistance to infection

3. the quantity of water ingested by bathers (related to time of immersion)

Humans are terrestrial animals designed to function in a dry environment. Immersion in water disrupts the natural defence mechanism of the body, which increases the chance of contracting an illness. Infection follows an upset in the host-parasite relationship (Cartwright, 1993), which means that swimmers are at a higher risk of illness independent of the microbial content of the water through submersion in an aquatic environment (Stevenson, 1953). Therefore, it is apparent that two points be considered when discussing illness acquired through bathing:

i. invasion of the body by a pathogen

ii. the weakening of the host by immersion in water.

Persons with a low immune system are open to attack by opportunistic micro-organisms. The infective dose of some viruses is thought to be as little as 1 particle (Fewtrell and Jones, 1992).

The principal infections potentially derived from seawater, listed in more detail by Phillip (1991) include gastroenteritis, jaundice, eye, ear, nose and throat infections, pneumonia, skin infections, salmonellosis, poliomyelitis, shigellosis, meningitis, and acute neurotoxicity. However, Fewtrell and Jones (1992) commented that most morbidity incurred through bathing is minor, and that occurrence of serious diseases such as cholera are minimal. In fact, although poliomyelitis and meningitis have been isolated from samples of sewage contaminated recreational waters, no substantiated occurrence of cases have been recorded in marine waters (Rees, 1993; Cabelli, 1983). Although contraction of disease from bathing in polluted waters might be minor, for example gastroenteritis (Cabelli *et al.,* 1982) or skin infections (Balarajan, 1992) the resultant effect often leads to discomfort, loss of leisure time and absence from work. However, it must be noted that the research discussed above (Brown *et al.,* 1987; Pike, 1994; Kay *et al.,* 1994) is based on circumstantial evidence, i.e. they only show a statistically significant relationship between bathing and an increase in illness amongst swimmers, and not a direct link. Proving a causative association between morbidity and bathing is much more difficult (Moore, 1975). An example highlighting this point can be observed through the two teenagers who contracted disabling symptoms following contact with the sea at Oxwich Bay (The Independent, 1994). Although medically both had symptoms believed by their physicians to be caused by ingestion of a water-borne virus, no viruses were detected in their blood making it impossible to retrospectively extrapolate back to derivation from the water (Wright, 1995).

3.4 Aetiological Agents

A wide range of pathogens are found in the marine environment. The most common which cause disease are bacteria, viruses, fungi and parasites (Melnick, 1984). Fewtrell and Jones (1992) give a comprehensive list of pathogens resident within UK marine waters. The prime source of pathogens believed to cause waterborne disease are derived from faeces and urine of warm-blooded animals (Knudson and Hartman, 1992). Enteric viruses have been proved to be a significant aetiological agent which occur in infected people (Berg and Metcalf, 1978), but are difficult to detect in water (Marzouk, 1984). It has been estimated that more than 100 types of pathogenic viruses occur in faecally polluted water and cause waterborne diseases (Melnick, 1984; Havelaar, 1993). Melnick (1984) noted the Norwalk RNA viruses in particular to be linked with waterborne disease, especially diarrhoea and gastroenteritis. A comprehensive table of viruses present in human excreta is presented by Melnick (1984). Cabelli (1981) and Havelaar (1993) supported this view by claiming that the Norwalk virus, the human rotavirus, the Hepatitis A virus, adenovirus and gastroenteritis viruses (such as the astrovirus) are likely to be the main aetiological agents of waterborne diseases. These range from mild gastroenteritis to severe meningitis. As yet no perfect indicator has yet been identified to accurately assimilate the presence of pathogens, although attempts have been made to model bacterial concentration against illness rates using dose-response curves (Cabelli *et al.,* 1982; Kay, *et al.,* 1994). For a detailed summary of epidemiological studies and resultant models describing health risk from bacterial indicator levels see Pruss (1996).

3.5 Theory of Indicators

Indicators are used to identify and quantify risk of exposure to pathogenic agents (Rees, 1993; Fleisher, 1990b). In the case of this research exposure relates to contact with faecal contamination of coastal waters. The WHO (1977) defines an health effect water quality indicator as an index of potential risk to health by a microbial, chemical or physical agent, substance or quality that arises from man's use of the aquatic environment for recreation or the production of food. Ideal indicators should demonstrate a high correlation with associated incidence of disease, based on sound scientific evidence (Kay, 1988). Selection of indicators must consider the environment to which the indicators will be exposed such as temperature, quantity of UV light, turbidity of the receiving waters and time of year (Pike, 1992). It is also important not to disregard the less tangible factors which must also be contemplated including economic, social and political aspects.

However, no indicator has yet been discovered which perfectly mimics pathogenic organisms (Berg, 1978). Potential indicators should fulfil the requirements laid out by WHO (1977):

a) be consistent and exclusively associated with the source of the pathogens, and occasionally, noxious substances;

b) be present in sufficient numbers or quantities without proliferation or somatic/genetic changes to provide a reasonable estimation of the presence of pathogens and the potential existence of a health risk;

c) approach the resistance to disinfection and environmental stress, including that resulting from toxic materials deposited in the aquatic environment, of the most resistant pathogen potentially present at significant levels in the source;

d) be quantifiable in environmental samples by reasonably easy and inexpensive methods, and with considerable accuracy, precision and specificity

Two methodological issues which need to be addressed when taking water samples and testing for indicators are variability of indicator over time and space and limited precision in techniques for indicator enumeration (Fleisher, 1990a; 1990b).

3.5.1 Bacterial Indicators of Sewage Pollution and Infective Disease

The considerable debate over appropriate indicators of microbiological content of seawater and the link to health risk from bathing is continuing (Lightfoot, 1989; HMSO, 1990b; WHO, 1991; Jones *et al.,* 1993; Pike, 1994). Historically, total coliforms, faecal coliforms and faecal streptococci have been the basis for most standards designed to ensure the health of bathers. Total coliforms have been shown to correlate with certain symptoms such as diarrhoea (Pike, 1994), but are currently regarded as being too abundant and ubiquitous in nature to be included as faecal indicators (ENDS, 1994d; EC, 1995). This has led to total coliforms being dropped from the proposal for the EC Bathing Water Directive (CEC, 1997). Faecal coliforms, primarily *E.coli,* are found only in mammalian guts and excreted in large numbers (Berg and Metcalf, 1978) making them the prime indicator of faecal pollution (Rees, 1993). The WHO (1991) have pinpointed *E.coli* as being the most effective index of faecally contaminated recreational waters. However, recent research suggests that faecal streptococci show higher resistance to environmental decay than *E.coli,* displaying a similar die-off rate similar to enteroviruses and other human pathogenic organisms (HMSO, 1990b), (Table 3.1). Faecal streptococci have also shown to be better correlated with water related illnesses (Kay *et al.,* 1994). In response the EC have reduced the Guideline parameter presently stipulated for faecal streptococci to 50 per 100ml and also introduced a Mandatory level of 100 per 100ml (CEC,1997). Criteria set in the Directive are for both marine and freshwater recreational sites. Mortality rates of bacteria vary depending on their environment and tend to be higher in saline waters than fresh waters (Cabelli *et al.,* 1982). Discharged bacteria, even faecal streptococci, might be undetectable in the environment within a few days of release, but enteric viruses might persist in an infective state at detectable levels for several months (Berg and Metcalf, 1978). Berg and Metcalf (1978) point out that absence of faecal indicator bacteria in samples of water or sewage does not guarantee absence of viruses.

Type	Harsh	Moderate+	Protected#
Coliforms	1-few hours	Hours-days	A few days
E.coli	Hours-1 day	A few days	Days-weeks
Faecal streptococci	1- a few days	Days-1-week	Weeks
Human pathogenic viruses	A few days	Days-weeks	Weeks-months

Table 3.1 Environmental Decay Rate of Micro-organisms in Marine Waters

(source: HMSO, 1990b)

* Sunny, warm clear seawater

\+ Deeper water, cooler, dull weather

\# Associated with suspended or settled sediment

3.5.2 Bacteriophage as Indicators of Sewage Pollution and Infective Disease

The emphasis on protection of health and amenity which provided the main initiatives to reform the existing EC Bathing Water Directive (CEC, 1976) placed greater importance on the enterovirus determinand of 0 per 10 litres. However, this is seen as unrealistic with enteroviruses being ubiquitous in the marine environment (HMSO, 1990b). The intention is to replace enteroviruses with a bacteriophage parameter once an appropriate one has been scientifically proven to be an adequate indicator of sewage (CEC, 1997).

Detection of viruses is a time consuming and expensive operation which requires well-trained personnel (Havelaar, 1990). The difficulties in selecting an appropriate bacterial indicator (Kay, *et al.,* 1994) capable of detecting pathogenic organisms, in particular viruses, has motivated the EC to look for alternative solutions. Bacteriophages (phages) are not new to the scientific community, and may provide an alternative solution (Stetler, 1984). Phages are viruses that attach to bacteria (Hugo, 1964; Scarpino, 1978). Many phages respond in a similar manner to human viruses in the environment and are relatively inexpensive and easy to detect potentially making them a viable substitute for enteroviruses (Fewtrell and Jones, 1992). The sampling process of identification is quick (Wentsel *et al.,* 1982). Research has shown that coliphages are highly correlated with total coliforms and faecal coliforms (Kennard and Valentine, 1974; Wentsel *et al.,* 1982; Jay, 1992). Stetler (1984) took this further by proving that coliphages have a higher correlation with enteroviruses than total coliforms, faecal coliforms and faecal streptococci. In support of selecting coliphages as an ideal indicator of sewage Kott *et al.,* (1974) showed that coliphages are found in significantly higher proportions than enteric viruses in wastewater (in the ratio of 3:1), very resistant to environmental decay and do not multiply in natural waters (Havelaar, 1993). Simkova and Cervenka (1981) also substantiate findings by Kott *et al.,* (1974) reporting enteroviruses and coliphage recovery rates to be similar in varying levels of pollution.

Morinigo *et al.,* (1992) suggested that coliphages could be used as an optimal indicator of micro-organisms in preference to the host bacterium *E.coli.* in accordance with findings from earlier work by Borrego *et al.* (1987). Although the coliphage family is large Havelaar (1993) has worked with the F-specific RNA phages and has identified them as being an appropriate indicator function of human pathogenic viruses in the water environment. Scarpino (1978) recognised the potential promise of bacteriophages as an index of viral pollution and suggested that further documentation of these micro-organisms be required before bacteriophages are promulgated as indicators of enteric viruses and enteric bacteria. Debate is still ongoing as to which, if any bacteriophage is selected for use as an indicator of pathogens in the aquatic environment.

3.5.3 Aesthetic Water Quality Indices

Current legislation governing quality of bathing waters focuses on identifying the physical, chemical and biological parameters (Moore, 1975; Cabelli *et al.*, 1982; Cabelli, 1983). Few studies address aesthetic quality of coastlines and the way in which visual pollution is perceived (David, 1971; Moser, 1984; Phillip, 1990), detailed in Chapter 5. Perception of coastal landscapes and in particular water pollution is an important criterion in quality assessment.

In the UK provision for water classification is set by the Water Act 1989, achieved through Statutory Water Quality Objectives (SWQOs) (NRA, 1991). SWQOs are water quality objectives set for inland water bodies, determined by type of use. As an alternative to current water classification schemes, work has been carried out investigating the suitability of indexing, attributing a single value to describe water quality (House and Ellis, 1980). Water quality indices reviewed for this study relate mostly to the physio-chemical and bacteriological quality of the water, but fail to acknowledge aesthetic factors. Other types of indicators used for water quality measurement include biological indicators, sensitive to particular substances. The Environment Agency use an indexing system to measure water quality of rivers based on benthic biological grading, dependent upon the relative abundance of macro-invertebrates (NRA, 1994). Another system uses dog whelks to measure levels of TBT, their relative numbers indicate levels of the toxin in water (Minchin *et al.,* 1997).

First attempts to design a water quality indexing system were made in the US by Horton (1965). Further research has been carried out developing an index rating scheme for water quality measurement through the 1970s (Moore *et al.,* 1970; Moore *et al.,* 1972; Joung *et al.,* 1979). Three main systems have been established in the UK. The Scottish Development Department (1976) worked on a system of weighting 10 variables and grouping them on a common scale to produce a single number between 0 and 100, which would reflect change in water condition over time. A value of zero would indicate crude sewage and a score of 100 would reflect excellent water quality. Ross (1977) also produced an indexing system which was primarily related to sanitary pollution. House and Tyson (1989) pointed to weaknesses within the water classification scheme developed by National Water Council (1977), such as subjectivity, reproducibility and sensitivity. In a study for North West Water Authority, they applied the Water Quality Index (WQI) developed by House (1986) to overcome these problems. Conclusions drawn showed that this system accounted for subjectivity within the methodology and produced a simple representation of results which were reproducible.

House and Ellis (1980) acknowledged that a Water Quality Index (WQI) is not totally objective, but does reduce a large volume of data to a single figure which is more readable and understandable to the general public. A major criticism of this type of measurement technique is the loss of information. In response to this House and Ellis

(1987) suggested using the WQI in conjunction with the existing NWC system of classification.

In recognising the need for a standardised approach required for operational management of surface water quality, House and Tyson (1989) implemented a schedule to investigate the effectiveness of the WQI, one of four water quality indices developed by House (1986). The WQI was formed around nine physio-chemical and biological parameters, which were converted to a 10-100 scale using specific rating curves and weighted similar to the system used by the SDD. Their findings demonstrated the WQI to:

- detect annual cycles and trends in surface water quality
- to highlight river reaches which have shown a change in quality over a specified time
- reflect both clean and polluted rivers alike and allow rivers to be placed into ranked order thus indicating spatial variations in water quality
- differentiate between rivers within the same NWC Class and indicate where class thresholds have been approached or crossed
- adequately reflect potential water use thus providing information to operational managers in terms of EC Directives for specific water uses
- assist in the evaluation of benefits to accrue from investment in capital schemes

As yet no defined index system for water quality appraisal exists. In addition, it appears that from the literature to date, work done is concerned only with streams, lakes and rivers (McClelland, 1973; Peterson, 1976). More importantly in the context of this work, there appears little acknowledgement of aesthetic values as a factor in calculation of water quality indices, or reference to the perception of the end user.

Water quality indices are a subsection of environmental quality indices (EQIs). Craik and Zube (1976, p.3) describing the term index as:

> *'The term index has usually been used in reference to an aggregation of individual indicators or measurements which collectively convey information about the quality of some complex aspect or component of a condition, property, or phenomenon'.* (Craik and Zube, 1976 p.3)

Craik and Zube (1973) acknowledged the visual and aesthetic importance in quality of scenic landscapes which physical indices fail to accommodate and comment on the significance of user based evaluations in designing EQIs.

When considering the coastal environment the dynamic nature of the sea and its tidal movement must be recognised, which varies internationally. However, common issues need to be addressed when talking about quality of water, whether it be river, estuarine or coastal. A sound methodological approach is necessary for developing a WQI. Daniel (1976) and Brush (1976) both identified specific needs for assessing the more global EQI. Daniel discussed the requirement to identify what constitutes environmental quality. He commented on the need for development of a weighting system which incorporates three factors, reliability, validity and utility and includes the less tangible human aspects to an indexing system. Brush (1976) also made clear the necessity to account for observer based perception in policy and development and management decisions.

Coughlin (1976) addressed specifically water quality and reviews research methods. He noted the multidimensional nature of water quality assessment and the complexity of evaluating water aspects which go beyond the realm of economics, such as moral issues. However, he questioned the ability of an index or single number to paint a complete picture of water quality. In discussing the use of a WQI, it is imperative that the way in which water quality is perceived be accounted for in any management decision making process which Coughlin (1976) acknowledged in agreement with Craik and Zube (1976), stating:

> *'Man's perception of water pollution is a central consideration in measuring water pollution. To rely on objective chemical measurements is not sufficient. The perception of water quality has a reality of its own which is just as valid, and perhaps of more importance in human decision-making than the reality of the measurement of physical and chemical properties'.*
> (Coughlin, 1976 p.206)

3.6 Epidemiology of Infective Disease from Bathing

Epidemiology is the branch of science which attempts to establish if specified hazards increases the risk of illness in exposed groups in comparison to non-exposed groups. In the context of this study exposure relates to contact with sewage contaminated seawater. Research into health effects associated with bathing date back to the early 1950s. Over the last two decades a substantial amount of investigations have taken place pushing the boundaries of knowledge further in this complex field (Moore, 1975; Mujeriego *et al.*, 1982; Brown *et al.*, 1987; Cheung, 1989; Alexander and Heaven, 1992; Von Schirnding *et al.*, 1993).

Four main study designs have been employed investigating health risk related to swimming and bacteriological quality of water (Pruss, 1996; Alexander and Heaven 1990).

3.6.1 Retrospective Cohort Studies

A retrospective cohort study begins with evidence of illness and attempts to work backwards to establish the possible cause. This type of design has inherent weaknesses in that the time lapse between recall of illness and activities is often long potentially leading to inaccurate information. In addition there is no control group and lack of information on bacteriological quality of the water makes it is impossible to establish if a dose-response relationship exists. An example of this type of study can be seen through the approach taken by the PHLS (1959) investigation into bathing water concentrating on the incidence of enteric fever and poliomyelitis.

3.6.2 Prospective Cohort Studies (Opportunistic Cohort)

The prospective cohort design was pioneered by Cabelli (1979), endorsed by the WHO (1989b) and further developed for the study commissioned by the DoE (Pike, 1994).

Information on populations or (cohorts) of bathers and non-bathers is collected at the beach along with personal details and history of illness. The groups are monitored using follow-up surveys either by post or telephone to investigate differences in levels of illness among the exposed and non-exposed groups and between levels of exposure. A main advantage of this type of design is that activities chosen on the beach by the participants are self-selected, overcoming ethical problems of having to survey on a beach which conforms to EC standards - see section 3.6.3 below. Another advantage is that this study type is applicable to the whole spectrum of age range, dependent only on those who decide to enter the water. In contrast the healthy volunteer study is restricted to adults aged over 18, as it would not be ethical to encourage children into recreational waters which could potentially effect their health. This approach lends itself to investigating the relationship between exposure and varying levels of water quality. In addition the WHO (1994a) recommended the prospective cohort study for local and low-cost application and where the sample group is relatively small. One main criticism of this type of study is that it relies on perceived or self-reported symptoms.

3.6.3 Controlled Clinical Cohort Studies (Healthy Volunteer)

The controlled cohort or health volunteer study was developed in Britain in 1989 and used in the DoE study (Jones *et al.*, 1993). Volunteers are recruited and randomised into bathers and non-bathers which maximises similarity between groups. Before exposure both groups are interviewed on the beach and given a medical examination. Following beach contact the participants are give a post-exposure interview and further medical examination. Accurate control of exposure to bacteriological levels is possible using this design and helps eliminate social and demographic bias. Medical examination aids in validating self-reported symptoms. The design type is limited as it may only be conducted on EC qualified beaches, and not those that fail for ethical reasons. As discussed above, only adults may participate aged over 18 may participate for ethical reasons. Further limitations are potential to induce interviewer/interviewee bias and financial and logistical costs of the operation are high (WHO, 1994a).

3.6.4 Cross-sectional Studies

The cross-sectional approach compares two groups of respondents, i.e. swimmers and non-swimmers against a particular agent thought to cause disease. The design requires respondents to be interviewed on the day at the beach and utilises a questionnaire to obtain relevant data. However, the participants in the study are not used in a post-beach interview, therefore the style takes no account of developed symptoms. An example of this type of study was conducted in the UK by Brown *et al.* (1987). Their study considered two beaches of different water quality and the findings showed swimmers who immersed their head to be the most susceptible to illness compared with a control group of non-bathers.

3.7 Epidemiological Research into Recreational Waters

The following literature review discusses the primary studies and findings from epidemiological/microbiological research into bathing since 1953.

3.7.1 American Work 1950s

The first major epidemiological study was carried out by Stevenson (1953) for the United States Public Health Service. Three locations were selected for the research study which recorded the swimming activities and occurrence of disease in a sample of bathers and non-bathers using a questionnaire. The main findings showed a higher incidence rate of illness among swimmers. Some correlation between bacterial content and illness rate was observed, in particular ear, eye, nose and throat infections, gastrointestinal problems and skin irritations. However, Stevenson concluded that this could be expected irrespective of the water quality, implying a lowering of the bodies immune system with exposure to water. He suggested that some of the strictest bacterial quality requirements for natural bathing water, then in existence, might be relaxed without significant detrimental effect on the health of bathers.

3.7.2 UK Work 1950s

The Public Health Laboratory Service (PHLS, 1959) conducted the first health effect study in the UK. Data was collected over a six year period on incidences of enteric fever (typhoid and paratyphoid) and poliomyelitis. The retrospective design chosen dealt only with major occurrence of disease. This approach requires extrapolating cases back to exposure. The results showed no cases of association between bathing water and poliomyelitis and only four cases of paratyphoid connected to sewage contaminated waters. The PHLS concluded that:

> *'..serious risk of contracting disease through bathing in sewage-polluted seawater is probably not incurred unless the water is so fouled as to be aesthetically revolting, public health requirement would seem to be reasonably met by a general policy of improving grossly insanitary bathing waters and of preventing so far as possible the pollution of bathing beaches with undisintegrated faecal matter during the bathing season'.* (PHLS, 1959 p.469)

The conclusions of both these studies were inconsistent, although different designs were employed. Results of the PHLS (1959) study provided the platform for the UKs stance on policy and lack of initiative concerning condition of its bathing waters until the mid-1980s (HMSO, 1984). It was not until implementation of the Water Act (HMSO, 1989) that recreational water quality was seriously considered with concerted effort to achieve European standards (CEC, 1976).

3.7.3 United States Environmental Protection Agency (USEPA) Research

Cabelli and various co-workers (1979; 1982; 1983) found the first credible dose-response relationship between disease and bacterial concentration. The study headed by Cabelli was conducted over a six year period between 1973-1978, sponsored by the United States Environmental Protection Agency (USEPA). Two marine waters and one brackish water were investigated. Cabelli *et al.* (1982) engineered the first prospective style beach survey using questionnaires and follow up surveys to investigate incidence of disease in respondents occurring within an incubation period of between 7-10 days, post survey day. A total of 25,442 participants took part in the research and a water sampling programme was followed on the trial days. Illness rate was compared between a control group consisting of all those who either did not enter the water or entered without immersing their head to those who did immerse their whilst performing their water activity. Disease was classified into two main categories: total gastrointestinal (TGI) which included all gastrointestinal symptoms and highly credible gastrointestinal symptoms (HCGI) which included either vomiting or diarrhoea accompanied by a fever, were disabling or caused either nausea or stomach-ache. The research team proved a statistically significant association between gastrointestinal symptoms and bathing related activities. A dose-response relationship between incidence of disease and levels of bacterial contamination proved significant. In particular a high correlation between illness rates and faecal streptococci was observed. Also elevated symptoms of illness were shown among bathers in even marginally polluted waters. The USEPA designed national US standards (USEPA, 1986) based on results of the Cabelli survey *et al.* (1982).

Cabelli's *et al.* (1982) work and the corresponding standards set by the USEPA have come under criticism, namely from Fleisher (1992) and Lightfoot (1989). Fleisher (1992) expressed concern about the pooling of data for both marine and brackish water (similar to freshwater), as indicators react differently under saline and fresh water conditions. In his paper he re-analysed the data set obtained by Cabelli *et al.* (1982) using multiple logistic regression. Results proved faecal streptococci to be predictive of gastrointestinal symptoms associated with marine waters, but not brackish water. Earlier work by Fleisher (1990a) questioned the USEPA's definition of a single maximum allowable mean faecal streptococci density to cover all marine bathing waters in the US (USEPA, 1983). Lightfoot (1989) criticised Cabelli's style survey, stating that his study measured perception of disease without supporting clinical evidence. Lightfoot (1989) made two other important points. First she questioned application of linear regression used to analyse the data, which does not take into account confounding factors and secondly that Cabelli classified swimmers as being only those who immersed their head, not accounting for other activities which can incur contact with water, such as wading, which potentially places people at risk.

Fleisher (1990a) commented on the lack of measurement precision used in the work by Cabelli *et al.,* (1982) in determining bacterial samples on the trial days stating the inherent variability of indicator organism over time and space. His criticism does not single out Cabelli's work but stated that a common failing of studies seeking to establish an association between swimming in recreational waters contaminated with domestic sewage was to adequately control for large amounts of measurement error contained in estimates of indicator organism densities.

3.7.4 World Health Organisation/United Nations Environment Programme Protocol

The prospective beach style approach pioneered by Cabelli and co-workers (*et al.,* 1982; 1983) was used in developing a protocol endorsed by the World Health Organisation (WHO) and the United Nations Environment Programme (UNEP) (1989b). The protocol was reconfirmed by WHO/UNEP (1991) and used in formulating guidelines for prospective microbiological/epidemiological studies for low-cost surveillance on health risks associated with recreational water use (WHO/UNEP, 1993). Development of the protocol based originally on Cabelli's work enabled international comparison of results and this approach has been used widely over the previous two decades (Brown, 1987; Alexander and Heaven, 1990).

3.7.5 Lightfoot - Developments in Statistical Techniques

Lightfoot (1989) carried out a prospective study of illness related to swimming activities at 6 freshwater sites in Southern Ontario during 1983. Data was collected on 6653 swimmers, 574 waders and 1193 people who did not enter the water. Water samples were taken and tested for specific bacteria and viruses. Results showed that the risk of disease was dependent on level of water activity. Crude morbidity data showed for overall illness the symptom rates were 76.8 per 1000 for swimmers, 41.8 per 1000 for waders and 19.3 per 1000 for those that did not enter. Apart from skin and allergic ailments, all illnesses were recorded at higher incidence for the those that entered the water. Lightfoot (1989) was one of the first researchers to employ the powerful statistical modelling technique, multiple logistic regression to her data. The conclusion from the modelling results showed that bathers were more likely to contract an illness than non-bathers, but no evidence was revealed to link disease and bacterial count.

3.7.6 UK Research

Major epidemiological studies have been conducted in the UK, the general consensus from the results found positive associations between bathing and increased risk of illness. Phillip *et al.* (1985) found a higher incidence rate of illness amongst swimmers who were snorkelling in Bristol docks, 27% of whom contracted gastrointestinal symptoms within 48 hours, compared to a control group of non-swimmers. Another study used a novel idea attempting to correlate pathogenic astroviruses (found in seawater) with water exposure (Croall, 1995). Astroviruses are believed to cause both mild symptomatic and asymptomatic illness in adults and children, although children are more susceptible (Croall, 1995). Results showed watersports enthusiasts to be twice as likely to have astroviruses in their blood stream in comparison to non-water users. Earlier work by Brown *et al.* (1987) and Alexander and Heaven (1990) used the Cabelli-style approach for their studies. Alexander and Heaven (1990) focused on children aged between 6 and 11 years of age using Blackpool Beach. Samples of seawater were taken simultaneously to the beach interview days, testing for total coliforms, faecal coliforms, faecal streptococci, salmonella and enteroviruses. Their results found a strong association between the children who were exposed to the water and a wide range of illnesses including gastrointestinal symptoms. Brown *et al.,* (1987) carried out work for Greenpeace over a six week period at two resorts in the South of England. Findings showed swimmers to be more likely to report symptoms of stomach upset, nausea and diarrhoea, the rate of which increased if head immersion occurred.

The Royal Commission on Environmental Pollution (1984) generally agreed with findings of the PHLS (1959) study regarding major illness, but was unsure about the risk of minor infection. An Advisory Committee set up by the DoE co-funded by the Department of Health, Welsh Office and NRA recommended that a series of epidemiological studies be conducted in the UK, to investigate health risks from bathing in sewage contaminated waters. The Water Research Centre (WRc) headed the investigation, based on the WHO/UNEP protocol Balarajan (1992). The study had two main objectives: i. to determine the extent to which health risk was related to bathing and ii. to evaluate if a dose-response relationship existed between risk to health and microbiological concentration in seawater (Pike, 1994).

The investigation used two design types, a controlled cohort study and a modified Cabelli-style beach survey (prospective cohort study) recommended by the House of Commons Environment Committee (HMSO, 1990b). A detailed discussion of epidemiological-microbiological design methods, is provided in section 3.6. The dual approach taken by the DoE was to capitalise on the advantages of each type of study. Investigations were carried out in three phases between 1989 and 1993.

3.7.6.1 Phase I of the DoE Study

A feasibility study using both a prospective cohort (Beach Survey) and controlled cohort study (Healthy Volunteer Survey) were employed at Langland Bay, Gower during August and September in 1989 (Pike, 1990). Both studies revealed higher reporting of symptoms among bathers than non-bathers. Total coliforms, *E.coli* and faecal streptococci were tested for and results indicated good water quality. No relationship was observed to exist between illness rates and bacterial concentration in the seawater.

The Beach Survey consisted of 4,045 subjects, of which 791 were contacted seven days later in the follow up survey. Increased reporting of ear and throat symptoms from bathers occurred over non-bathers and higher levels of water activity resulted in increased levels of self-reported illness (Balarajan *et al.,* 1991). The Healthy Volunteer study consisted of 276 completed responses from an initial 465 participants recruited.

Findings of the Healthy Volunteer study were similar to the Beach Survey, with bathers reporting higher rates illness over non-bathers with particular reference to ear, eye, throat and gastrointestinal problems (Jones *et al.,* 1991).

3.7.6.2 Phase II of the DoE Study

Phase II was based on the pilot studies conducted at Langland Bay. Both the Beach Survey and Healthy Volunteer study were believed to be suitable methods for investigating health effects from bathing, but much larger numbers were recommended to reach statistical significance (HMSO, 1990b). Ramsgate, Kent was the site chosen for the Beach Survey and 1883 subjects were successfully contacted by telephone, post-beach interview (Balarajan *et al.,* 1991). The Healthy Volunteer study was conducted at Moreton, Merseyside and achieved a post-exposure response of 303 (Pike, 1991). Results of both designs found swimmers to record higher rates of illness than non-swimmers confirming the findings from Phase I. The Beach Survey also showed children who bathed to be more susceptible to illness than older people who bathed and a correlation between morbidity rates and bacterial concentration was noted (Pike, 1991). Balarajan *et al.* (1991) stated that results of the Beach Survey proved the applicability of the protocol endorsed by WHO/UNEP (1989b), and further developed by the WRc, to UK waters.

3.7.6.3 Phase III of the DoE Study

Results of the pilot studies and phase II studies motivated the Department of the Environment to invest in further research into health risks from bathing; the results also provided the baseline to derive optimum sample sizes to achieve statistical significance (HMSO, 1990b). Both the Beach Survey and Health Volunteer survey designs were utilised for the Phase III studies, and conducted during 1991 and 1992.

The first half of the Beach Surveys were carried out at Paignton, Lyme Regis, Rhyl and Morecambe in 1991 (Balarajan, 1992). The second half of the Beach Surveys were carried out at Cleethorpes, Skegness, Instow and Westward Ho! in 1992 (Balarajan, 1993). The aim was to achieve 2000 subjects per beach. The beaches were chosen to demonstrate a wide range of water quality ranging from very clean, just passing and failure in line with the EC Bathing Water Directive (CEC, 1976). The general conclusion from results obtained from the eight beaches showed a dose-response effect in the risk of reporting symptoms with increasing levels of seawater activity and risk of reporting symptoms was higher at beaches with poorer water quality. In particular there was clear incidence of increased reporting of gastrointestinal and diarrhoea amongst those that entered the water (Balarajan, 1993).

The Healthy Volunteer study phase III was also conducted in two halves. The first study was carried out at Southsea, 1991 and the second study was at Southend on Sea, 1992 (Jones *et al.,* 1993). Volunteers were randomised into two groups, bathers and non bathers and all volunteers were provided with packed lunches. Water quality at the beaches ranged from excellent to marginal. Results revealed higher incidence of gastrointestinal illness in the exposed group vs. the control group of non bathers. There was also increased rates of ear infections, sore throats and skin symptoms amongst the exposed group. The only indicator to show a dose-response relationship with illness was faecal streptococci at chest depth, at concentrations exceeding 35 per 100ml (Jones *et al.,* 1993).

3.7.6.4 Summary of the DoE Studies

The combined results of both the Beach Surveys and Healthy Volunteer studies into health effects of sea bathing created a lot of controversy (ENDS, 1994f). The final report concentrated mainly on the Beach Surveys (Pike, 1994). Both study designs at the respective beaches revealed increased rates of illness in the exposed group in contrast to the non exposed group, in particular gastrointestinal, ear symptoms, sore throats and skin symptoms. In addition correlations between illness rates in swimmers and bacterial concentrations were noted. The Healthy Volunteer study showed faecal streptococci to elevate the illness rates once the concentration exceeded 35 per 100ml and the Beach Surveys found higher levels of activity to increase the rate of self-reported symptoms (Pike, 1994). These results confirmed findings of previous studies (Cabelli, 1983; Lightfoot, 1989; Alexander and Heaven, 1990) which added weight to the proposed amendments to the EC Bathing Water Directive (CEC, 1976). ENDS (1994c) reported that the implication of changes to the Bathing Water Directive would almost certainly increase the number of British beaches failing to meet Mandatory standards.

The UK Government refuted the evidence of findings produced by both the Beach Surveys and Healthy Volunteer studies and claimed they lacked statistical significance, in partial agreement with Pike (1994). The DoE commissioned the WRc to re-analyse the data to further investigate the relationship between bacterial indicator density and reporting of water-related symptoms (WRc, 1996b). The results proved no positive relationship between rates of illness and concentration of total coliforms, *E.coli* or faecal streptococci. The modelling process controlled for non-water related factors, which included age, visitor type, and consumption of particular foods (similar with this thesis). In addition to the linear logistic modelling of the Beach Survey data two additional models were applied, a generalised non-linear logistic model and a generalised linear model without logistic transformation. Both models were of limited value due to difficulties in fitting the data

The final report to the DoE (WRc, 1996a) re-affirmed the WRc final report phase III (Pike, 1994) which found both the Beach Surveys and Healthy Volunteer studies agreed on increased illness rates from exposure to seawater and higher levels of activity. However, it also stated that the increase in reporting of symptoms was irrespective of whether water quality was good or bad in relation to EC Mandatory standards (CEC, 1976). This supported the view that the predictive model produced by Jones *et al.* (1993) was insubstantial and re-stated the lack of evidence found between indicator density and illness rates from the second analysis of the Beach Survey data. In view of findings of both sets of studies, the WRc concluded that tightening the EC standard on bathing water would be mis-leading to the general public by inferring they would receive greater health protection (WRc, 1996a). Rees (1994) was in opposition to this approach stating that lack of statistical evidence was insufficient to keep current standards in light of clear evidence of relating swimming and disease. Since the WRc reports (1996a; 1996b) have been published, the EC have implemented the reforms to the Bathing Water Directive (CEC, 1997).

3.8 Setting of Standards

It is apparent from the literature that there is a definite health risk from bathing in sewage contaminated waters (Cabelli, 1983; Lightfoot, 1989; Jones *et al.,* 1993), even where water quality meets European Guideline standards (Phillip *et al.,* 1985; Alexander and Heaven, 1990; Balarajan *et al.,* 1991). The House of Commons Select Committee for the Environment (HMSO, 1990b) commented on the difficulty in creating standards for water quality based on scientific criteria. Derivation of health based standards are difficult (WHO, 1994c) and more specifically setting an objective number for a water quality indicator(s) that represented an acceptable risk to health is problematic (Geldreich, 1970; Rees, 1994;). The WRc (1996a) outlined four methods to achieve reasonable standards. They were in brief:

Attainment - to arbitrarily set water quality standards based on available technology and not related to levels of pollution. This approach utilises available technology, but gives no guideline on what overall total expenditure should be or level of health risk from swimming.

Detectable risk - to relate the concentration of pollution which induces a detectable level of health risk. In theory this method holds water, but in practice it is dependable on the validity of the studies used to detect the risk and determining a precise threshold would be difficult to achieve with environmental conditions varying widely across different recreational waters.

Acceptable risk - to set a standard that forms a boundary of acceptable risk. Under these conditions those who bathe would take a known risk of acquiring one or a number of specific illnesses. In pragmatic terms defining this level is dependent on the information between illness and quality of recreational waters, which range considerably under diverse environmental conditions. Selection of the discriminatory standard would also be complex determining an acceptable level or risk.

Cost-benefit analysis - this involves a trade-off between engineering costs to improve the water quality against benefits of swimming in a cleaner sea. This approach has a few pitfalls. First even thought the cost is carried by the general public they are not evenly distributed and secondly it is virtually impossible to achieve a consensus about the way sickness and health should be traded.

There has been a lot of criticism over standards set by the EC Bathing Water Directive (CEC, 1976), and the monitoring regime used (Brown *et al.,* 1987; Mujeriego, 1988; Fleisher, 1990a; HMSO, 1990b), (refer Section 3.9.2.6). Claims have been made that the directive was based on limited epidemiological evidence (Kay, 1988; HMSO, 1995b), highlighting the controversy which exists amongst the scientific and political communities over the setting of appropriate water quality standards. Wheeler (1990) acknowledged this, stating there are a number of criteria that standards can be based on, such as ecological, medical, political, and economic each having its own set of dimensions and scientific justification. Pike (1994) noted that a continuous relationship exists between health and water quality, which made the design of acceptable risk levels difficult (Rees, 1993). Instead of the current European pass or fail rule, Wheeler (1990) suggested a water classification system as an alternative for defensible health criteria for recreational water quality standards. Findings by Kay *et al.* (1994) from the controlled cohort studies also accounted for the continuous relationship between health and concentration of microbiological indices. This gave a precise method of calculating risks of contracting gastrointestinal from bathing in waters of varying levels of bacterial pollution (ENDS, 1994b). The complexity of establishing health related standards for water quality was acknowledged by Grantham (1992) who stated that they are reliant upon:

- The health of the community served by the local discharge(s)
- The bathers resistance to infection
- The quantity of water ingested by bathers (related to time of immersion)

The European Commission responded to calls for a revision of the Bathing Water Directive and came up with a set of reforms (CEC, 1994) which have now been re-amended and confirmed (CEC, 1997). However, there has been contention over these amendments, mainly the inclusion of a Mandatory standard for faecal streptococci of 400/100ml which has been further reduced to 100 per 100 ml, and more emphasis being place on the enterovirus parameter of 0 per 10 litres (refer Section 3.9.3). At the House of Lords Select Committee on the European Communities (HMSO, 1995b) Hilton (DoH) questioned the statistical significance of the faecal streptococci level being proposed by the EC on the grounds that it has been founded on research reliant upon self-reported symptoms. Hilton also argued that results of the cohort study done by the WRc (Jones *et al.,* 1993) showed only a weak epidemiological association between exposure and symptoms and there was no proof to suggest these findings would be applicable to other sites. However, the Committee (HMSO, 1995b) opposed this view, rather calling into question whether the faecal streptococci level was stringent enough based on results produced by Kay *et al.* (1994). If the current level outlined in the proposed directive remains the likelihood will be an increase in numbers of British bathing waters failing to meet EC standards (CEC, 1976) and the cost incurred to ensure compliance would be *circa* £1 billion (ENDS, 1994a).

In contrast North America bathing water standards are much more stringent than European standards (Nelson *et al.,* in press (a)). If USEPA or Toronto standards were applicable to UK waters a substantial elevation in non compliance would occur. The monitoring procedures used in North America are also well in advance of the system used in Europe. In Toronto for example daily samples are taken, and if the 10 day geometric mean exceeds 100 faecal coliforms per 100ml the beach is closed. The EC monitoring framework relies on retrospective grading of beaches based on the previous years results, failing to give up to date information (Kay *et al.,* 1990). Nelson *et al.,* (in press (a)) compared European and North American water quality standards displayed in Tables 3.2 and 3.3 highlighting respective monitoring differences.

Cartwright (1993) recommended that over the next decade more emphasis should be placed on understanding the pathogenesis of disease in more detail so that appropriate control measures be undertaken. The view was supported by Rees (1993) who acknowledged the necessity to set rational standards to ensure the protection of health and the environment based on sound epidemiological studies, independent of political constraints and spurious cost-benefit analyses.

Agency	Regime	Faecal coliforms standard
Toronto Health Department	Daily	GM < 100 ml^{-1} No sample to exceed 400.ml^{-1}
Canadian Federal	5/30 Days	GM < 200 ml^{-1} Resample if any sample exceeds 400 ml^{-1}
U.S.E.P.A.	5/30 Days	GM < 200 ml^{-1} < 10% only to exceed 400 ml^{-1}

Table 3.2 North American Bathing Water Quality Standards

Fortnightly sampling	**T.coliform 100ml^{-1}**	**E.coli 100ml^{-1}**	**F.streps 100ml^{-1}**
EC 76/160 **Current standard**			
Imperative level 95% of samples should not exceed this figure	10,000	2000	--
Guide level 80% of samples should not exceed this figure	500	100	100
EC Com(97) 585 Final **Amendments**			
Imperative level 95% of samples should not exceed this figure	--	2000	100
Guide level 80% of samples should not exceed this figure	--	100	500

Table 3.3 European Bathing Water Quality Standards

GM = geometric mean (average value of a set of n numbers expressed as the n^{th} root of their products).

(source:cited Kay *et al.,* 1992, page 14)

3.9 History of Water Quality Legislation

Standards for bathing waters are set by the EC, but before reviewing the Bathing Water Directive in full, it would be useful at this stage to discuss the formation and development of UK water quality legislation and the framework within which it works. The first piece of legislation directly addressing the issue of aquatic pollution was drawn up in 1876, making pollution of rivers an offence (Haigh, 1995a). It was not until 1951 that a more up to date version appeared in the form of the Rivers Pollution Prevention Act (1951), dealing primarily with the sustained quality of rivers and inland waters, which was extended much later to cover some estuaries and tidal waters by the Clean Rivers (Estuaries and Tidal Waters) Act 1960 (Haigh, 1995a). These two Acts placed greater responsibility on River Boards to maintain the quality of rivers and also gave them power to grant discharge consents and emission standards for disposal of waste. Although water quality legislation in the UK dates back to the beginning of the century, the first epidemiological study was not commissioned until the 1953 by the Public Health Laboratory Service (PHLS) (refer Section 3.7.2). Up until the mid-eighties the UK government relied on the results of this study to justify minimal improvements to bathing waters (HMSO, 1984). The general conclusion from the PHLS stated:

> *'Bathing in sewage-polluted sea water carries only a negligible risk to health, even on beaches that are aesthetically very unsatisfactory'.*
> (PHLS, 1959 p.468)

The two next major steps in the legislative evolution process were the Water Act (1973) and 1974 Control of Pollution Act (COPA 1974), which created a policy environment conducive to effective water quality management (House and Ellis, 1980). However, it was not fully enacted until 1986. Haigh (1995a) commented on the COPA 1974, claiming that the only original feature was to provide the public's access to information about discharges, whilst otherwise re-enforcing earlier Acts and extending its range to cover discharges to waters not previously controlled.

3.9.1 The Water Act 1989 and Environmental Protection Act 1990

Implementation of the Water Act 1989 (HMSO,1989), consolidated by the Water Resources Act 1991 (HMSO, 1991) formed the structure of the privatisation of the water industries and creation of the National Rivers Authority (NRA) 1989 (Vaughan, 1993). This defined the duties of the private-sector water service companies as responsible for drinking water supply and the sewage process. The newly formed NRA were made the 'competent authority' (NRA 1991) endowed with a wide range of duties including the monitoring and control of water pollution (Grantham, 1992). Section 85 of the Water Resources Act (HMSO, 1991) is often used by the NRA in prosecution. The Act make it an offence to knowingly permit poisonous, noxious or polluting matter or solid waste matter to enter 'controlled waters'. Haigh (1995a) stated that this formally separated the functions of the water industry as polluter and regulator. The introduction of the Water Act (HMSO, 1989) coupled with provisions made in the Environmental Protection Act formulated in 1991, provided the main legislative powers controlling water pollution and conforming to European legislation regarding quality of recreational waters.

The Water Act (HMSO, 1989) gave powers to the Secretary of States for Wales and Scotland and DoE for England to create water quality objectives for controlled waters. The NRA are the responsible agency for instituting this policy. To ensure compliance with EC legislation a formal system of statutory water quality objectives (SWQOs) were introduced (NRA, 1992a). SWQOs are a further development of the National Water Classification system (NWC,1977) which established criteria for classifying waters on the basis of particular resource use (NRA, 1992a). All types of water are included, e.g., rivers, lakes and groundwater, to incorporate the needs of relevant European Commission Directives (Green and Birchmore, 1993). The Secretary of State, following consultation with the Environment Agency and other appropriate agencies is responsible for determining what SWQO is to be set for a particular water and by a specified date (Howarth, 1992). The Environment Agency have to then ensure that the respective SWQO is achieved by reviewing individual discharge consents. However, Earll (1994) believes that SWQOs for estuaries and coastal waters will not be established for many years. There has been criticism over this system by environmental groups due to the parameters prescribing the SWQOs in chemical terms only, without biological reference.

The two main driving forces to clean up the quality of water at beaches, rivers and other inland waterways are the European Directive concerning the quality of bathing water (CEC, 1976), since reformed (CEC, 1997), and the European Directive concerning urban waste water treatment (CEC, 1991). Although the Directive concerning bathing water is of prime importance it is not the intention of the EC that it be seen in isolation from a body of legislation aimed at maintaining and improving the aquatic environment (Howarth 1992). Other relevant Directives concerning the aquatic environment include the EC Directive on the quality required for shellfish waters, (CEC, 1979), the Directive on pollution caused by certain dangerous substances discharged into the aquatic environment of the Community, (CEC, 1976b) and also the Directive - Committee Decision setting up an advisory committee on oil pollution. For a review of both European and UK legislation regarding water quality see Haigh (1995b).

In addition to water quality regulation, EU Member States are advised by other European initiatives and international organisations and agreements to improve the environment. For example the European Charter on Environment and Health (WHO, 1989) highlights the importance of health to Member citizens recognising the environment as a resource for well-being incorporating physical, psychological, social and aesthetic aspects. The WHO produce documents for environment issues, including water quality guidelines (WHO, 1993). A major breakthrough came at the Earth Summit in Rio when an international collective of governments sat down to create Local Agenda 21 (Harman *et al.,* 1996) focusing on a combined approach to encourage local authorities to practice sustainable management plans for the environment, which inevitably apply to the coast.

3.9.2 European Council Directive Concerning the Quality of Bathing Water

3.9.2.1 Formation and Intentions

EC Directive concerning quality of bathing water (CEC, 1976) was the first European initiative directed at improving the quality of waters for recreational use among Member States (see Appendix IV). The purpose of the Directive was that the quality of bathing water was to be raised over time, or maintained, not just to protect public health but also for reasons of amenity. This was to be done largely by ensuring that sewage was not present or had been adequately diluted or destroyed before discharge to the aquatic environment (Haigh, 1995b). The Directive, formulated in 1976, which has since been reformed (CEC, 1997), is a legally binding instrument. Although the parameters outlined in the Directive have come under extensive criticism, the Directive created the motivation for Member States to begin cleaning up their recreational waters, stating in the preamble the requirement:

> *'to protect the environment and public health, it is necessary to reduce the pollution of bathing water and to protect such water against further deterioration'.*
>
> (CEC, 1976 p.81)

Vincent (1992) stated that the principal obligation of the Directive is that Member States ensure that the quality of bathing water meets the Directive's standards (Article 4), and that the rest of the Directive is concerned with defining this obligation. He further commented that perhaps the most significant sentence in the Directive is that which comprises Article 6(3), which states:

'Local investigation of the conditions prevailing upstream in the case of fresh running water, and of the ambient conditions in the case of fresh still water and sea water should be carried out scrupulously and repeated periodically in order to obtain geographical and topographical data and to determine the volume and nature of all polluting and potentially polluting discharges and their effects according to the distance from the bathing area'

(CEC, 1976 p.83)

This contextualises the role of the competent authority in identifying sources of pollution and taking action to protect bathing water quality. In setting standards defining acceptable water quality for bathing, the Directive sets out a list of bacteriological and physio-chemical parameters with which each Member State must ensure compliance at identified bathing waters.

3.9.2.2 Technical issues-

The parameters originally set in the EC Bathing Water Directive (CEC, 1976) have been subject to much discussion and criticism (Wheeler 1990; NRA 1991b; Grantham, 1992). Nineteen physical, chemical and microbiological parameters were constructed to obtain the objectives of the Directive, which included tests for pH, colour, mineral oils, surface active substances, phenols and transparencies with thirteen having both an Imperative or Mandatory standard (I) and a more strict Guideline standard (G). A degree of flexibility was given to Member States in implementing the Directive and selection of parameters to test. The most important of these values selected to determine bathing water quality were the microbiological parameters total coliforms and escherichia coliforms (*E.coli*) (ENDS, 1994g), which are bacteria used as an indicator of sewage rather than tested as a pathogen. *E.coli* in particular is only associated with mammalian guts and excreted in vast numbers (Rees 1993b), and its ubiquitous nature in the marine environment coupled with relative ease of analysis consequently makes it a common indicator of faecal pollution (WHO/UNEP, 1991). A number of parameters which included enteroviruses, salmonella and faecal streptococci were defined and only needed to be checked for in recreational waters when there was a deterioration in quality or if a problem was believed to exist. The implication was that these determinands do not have to be checked regularly at all sites (See reforms to the Directive, Section 3.9.3). The Directive made provision for waivers under exceptional weather, geographical conditions or natural enrichment regarding certain parameters including colour and transparency (NRA, 1991b).

Standards laid down were based on limited epidemiological science, with insufficient information on selection of appropriate indicators available (Kay, 1988). For example, *E.coli* has been proved to show little correlation with swimmer illness rates compared to non swimmers, especially gastro-intestinal symptoms (Cabelli *et al.,* 1982; Kay *et al.,* 1994). Grantham (1992) noted that this flaw in the Directive highlights the emphasis placed on amenity value, irrespective of health risk. Viruses are responsible in most cases of causing morbidity in bathers (Berg, 1978; Melnick, 1984; Walker, 1992). However, viruses are far more resilient to environmental stress than coliform bacteria, and can last up to three week in marine waters (Berg and Metcalf, 1978). Without full sewage treatment, low detection of coliform bacteria does not mean low viral contamination (Berg and Metcalf, 1978). The Directive made provision for testing faecal streptococci, now recognised as a better indicator of sewage and pathogenic organisms because it is more resistant to decay than the coliforms and therefore a better indicator of viruses (Kay & McDonald 1986; Kay *et al.,* 1994). The original Bathing Water Directive (CEC, 1976) created only a Guideline standard for faecal streptococci, enabling Member States to eliminate this parameter from testing their bathing waters.

3.9.2.3 Compliance and Litigation

The standards originally set by the EC Bathing Water Directive in 1976 had to be complied with by Member States, through the establishment of national laws and creation of administrative structures. In contextual terms the Directive defined Member

States obligation to meeting the standards as '*binding as to the result to be achieved ... but shall leave to the national authorities the choice form and methods*'. The form of this statement explicitly stated the requirement of respective States to ensure that the objectives of the Directive were met at designated recreational water sites, but there was a degree of flexibility allowed in choice of methods selected.

The EC has the authority to enforce Community legislation on Member States that fail to comply with any Directive, which usually results in the particular State having the opportunity to take remedial action to correct the situation. Implementation of the Directive in the UK was enhanced with the establishment of the Water Act 1989, (refer Section 3.9.1), which gave the NRA empowerment for guarding the quality of recreational waters acting as the Government watchdog. The Bathing Water Directive (CEC, 1976) initially established a time period extending to 1985 for each State to ensure that designated bathing areas reached Directive standards, although derogations were granted under exceptional circumstances based on plans for the management of water within the area concerned, (Article 4). The Directive formulated an objective definition of bathing water, (Art.2(a)):

> '*all running or still fresh waters or parts thereof and sea water, in which: bathing is explicitly authorised by the competent authority of each member State, or bathing is not prohibited and is traditionally practised by a large number of bathers*'.
> (CEC, 1976 p.82)

A large number of UK recreational waters failed to comply with values set by the Directive. The British Government tried to extend the compliance date and attempted to avoid prosecution by only identifying bathing waters which would meet the necessary standards. Initially the UK only designated 27 bathing waters, which showed poor regard for the Directive. The Government relied on the principles that there were no bathing areas where bathing was explicitly authorised, and used the lack of clarity stated in part 2 of the article which defines a bathing area as being traditionally practised by a large number of bathers to limit defining its designated waters. The UK Government ruled out inland waterways due the lack of use by large number of bathers and concentrated on coastal waters, but took advantage of the fact that many beaches are utilised by recreationalists who do not enter the water.

The EC rarely impose obligations against individuals (Howarth, 1992) and would only bring legal proceedings against a State to the Court of Justice of the European Communities as a last resort, and would only be made when no other option remained (Vincent, 1992). However, persistent breach of the Directive standards can result with infraction proceedings taken against the respective Member State, as the UK found out when they were brought before the European Court of Justice in July 1993 (Case C-56/90-Rees, 1994) for not identifying popular tourist beaches. The Court concluded that the UK had failed in their obligation to meet the standards set by the Bathing Water Directive at particular bathing sites, including Blackpool and Southport.

The result of the prosecution led the UK Government to review their criteria for identifying bathing waters in 1987, and significantly more were included. At that time only 56% of the designated beaches reached the standards set by the Directive. In 1989 the Government in consultation with the water authorities announced that it would invest £1.4 billion pounds (ENDS, 1993) to install long sea outfalls at coastal towns in order to disperse the pollution out at sea. The intention was that 95% of UK bathing waters would comply by 1995. Then with pressure from the awaited urban waste water treatment Directive a further expenditure of £1.5 billion pounds (Rees 1993) was then announced 4 months later to introduce new primary treatment systems. The investment substantially increased the number of bathing waters meeting the EC Mandatory standards (CEC, 1976). By 1996 the figure of compliance was up to 87.5% (NRA, 1996). Presently the UK has 464 identified bathing waters, although none include freshwaters, in contrast to other EC countries. France in comparison have 1362 inland waters identified as recreational bathing areas (NRA, 1991b).

3.9.2.4 Sampling procedures

The Environment Agency is responsible for sampling designated water sites in the UK. They take 20 samples during the bathing season between May 15 and September 30 in England and Wales and between June 1 and September 15 in Scotland and Ireland (NRA 1991b). The Directive allowed a reduction in the sampling frequency of 50% if results of water quality were appreciably better than the Directive standards for the previous year (CEC, 1976). Certain physical parameters are checked visually, the microbiological determinands require analytical techniques. The Directive allowed flexibility in determining methods of analysis for coliforms and faecal streptococci, quoting the acceptability of either the multiple tube fermentation with a most probable number (MPN) or membrane filtration technique (refer Section 6.1.2.1).

Compliance was based around 95% of samples passing Imperative standards. Guideline standards had a reduced compliance rate of 80%, these included faecal streptococci. Results of all tests are issued to relevant local authorities who are responsible for displaying the data at the sampled site. The intention being that the public can make an informed decision on whether to bathe at the particular beach.

3.9.2.5 Application in England and Wales

Although local authorities in England and Wales have the responsibility of displaying water quality test results at identified beaches, very few have comprehensive coastal zone management policies. In most areas the respective authority tend to concentrate on safety, involving lifeguard cover and daily beach clean-up operations. Local authorities appear to regard tourism and economic aspects of beach management as paramount on their agenda, without making the obvious connection to water quality at their bathing beaches, irrespective of the duty to protect the health of bathers. Two examples, with jurisdiction over the areas covered by this study include the Vale of Glamorgan and Swansea City Council. The Vale of Glamorgan have no specific policies towards water quality or the health of bathers at their beaches (VOG *pers.comm.*, 1996b). Swansea City

Council have a management plan for Gower peninsular (Mullard *pers.comm.,* 1996*).* This is primarily land based although a coastal study was undertaken on Gower in conjunction with Swansea City Council investigating management of water sports and local conservation issues (Nelson, 1994.

Although the Bathing Water Directive applies to all Member States, problems have arisen over the accuracy of international comparison of results between EC countries. The UK face further problems in achieving compliance in contrast to other Member States due to the UK being the only country that consistently manages to successfully monitor all of its designated bathing waters (Robens Institute 1993). Both the Marine Conservation Society (1994) and a report by the Government (HMSO, 1990b) have similarly expressed their concerns over inadequate sampling programmes in other countries.

The UK have failed to designate inland waterways under the Bathing Water Directive, mentioned above, which often receive heavy recreational use including windsurfing and sailing. In addition lower cost more effective wetsuits are increasing surfing participation (Surfers Against Sewage, 1995), which is an all year round sport. In light of these facts there is a need to accommodate changing trends in water sports and to expand the sampling regime over 12 months encompassing inland waterways to aid in assessing and controlling health risks for all water recreational use.

3.9.2.6 Criticisms of the Directive

Wide spread criticism has been directed at the EC Bathing Water Directive (Vaughan, 1993; HMSO 1990b). The original Directive (CEC, 1976) is now over twenty years old, and antiquated in light of more current studies (Cabelli *et al.,* 1982; Pike, 1994). As stated earlier the water quality standards were based on limited epidemiological evidence (Kay, 1988), and in particular have little public health significance to coastal bathing waters in temperate north-west Europe (Kay *et al.,* 1994). The House of Lords Select Committee (HMSO, 1995b) noted that there was no publicised rationale for selection of standards, a view backed by Cartwright (1992) who commented that no information on derivation of standards were laid down or proof of a relationship between compliance and health risks from bathing.

The main criticism of the Directive has been the selection of inappropriate bacterial indicators, a problem faced by most epidemiological studies. Total coliforms are not exclusively of the mammalian gut and therefore not believed to be directly related to health risk from bathing (Dutka, 1973; Phillip, 1991; ENDS, 1994). Emphasis has shifted to inclusion of a Mandatory level for faecal streptococci into the reformed Directive (CEC, 1997), which originally only had a Guideline parameter attributed to it. Faecal streptococci has proved a better indicator of health risk to swimmers than both total coliforms and *E.coli* (Kay & McDonald, 1986). Determinands for enteroviruses and salmonellae have also come under criticism (Section 3.9.3.1) due to unachievable standards, zero per 10 litres and zero per 1 litre respectively. It is generally thought these are too stringent, especially as enteroviruses are ubiquitous in the marine environment (HMSO 1990b; Grantham, 1992).

Questions have also been aimed at sampling frequency required during the bathing season. The UK for example is only obliged to take a minimum of 12 samples per year. The Environment Agency take 20 samples during the bathing season May to September, which ultimately means two failures would mean non compliance. Such a small sampling frequency only provides a snapshot of the water quality, without accounting for temporal, spatial or tidal variations, making statistical interpretation difficult (Jones *et al.,* 1990; Grantham 1992). Assessing compliance in a technical sense has been another cause for concern. The flexibility given to choice of microbiological analysis used and the high variability of environmental conditions at the sampling site, mean that it is very difficult to compare results between neighbouring countries within the Community (Rees 1994). Grantham (1992) similarly noted that compliance of bathing waters might well depend on type of analysis selected, which is in addition to concern over actual reliability of sampling techniques (Fleisher, 1990a).

Further to the discussion over appropriate parameters outlined above the bacteriological requirements of the Directive are less rigorous than those set in the United States and Canada (Balarajan, 1992). Finally, the Directive concerns itself with long term management of bathing waters and not day to day control of health risks (NRA 1990). The retrospective approach taken means determination of water quality is reliant on the previous years results. Compliance is either pass or fail, which in itself is flawed. A continuous relationship between health and water quality exists and cannot be based on a cut off point (Pike, 1994), although definition of acceptable levels of risk are difficult (Rees, 1993).

3.9.3 Amendments to the European Bathing Water Directive

Increased evidence of risk to bathers from swimming in sewage contaminated waters (Robens Institute, 1987; Pike, 1990; Balarajan 1992; Jones *et al.,* 1993) and questions over the applicability of the EC Bathing Water Directive set in 1976 (CEC, 1976), gradually applied pressure to the European Commission to review the parameters laid down (Wheeler, 1990). In 1994 the EC announced proposed reforms to the Bathing Water Directive (CEC, 1994), with the main drive placed on protection of health ensuring bathers throughout the community receive a guaranteed minimum level of protection (ENDS, 1994c). Focus for the amendments were derived in light of improved scientific knowledge concentrating on revamping existing legislation, but also making it simpler in its execution (ENDS, 1994a). The proposed amendment has now been slightly altered and enforced from 31 December, 1997 (CEC, 1997). The points on protection of health and acknowledgement of scientific advancement are highlighted in the preamble:

> *'....the list of parameters to be measured should indicate in the most appropriate way the quality of bathing water and take account of advances in science and technology; whereas there is need to require the verification of only those parameters which are indispensable for ensuring an adequate protection of human health'*
>
> (CEC, 1997 p.2)

An explanatory memorandum attached to the Directive makes clear the intention to simplify the standards without Member States incurring any extra expense, phrasing the change as being a 'neutral translation'. It is imperative that serious consideration is given to the essential core criteria, and underpinned by current scientific understanding. Rees (1994) noted hazards, acceptable risk, definition of accurate and monitorable indicators of risk, and set standards that are both achievable and enforceable to be important aspects of regulation regarding bathing waters.

3.9.3.1 Determinands

With reviewed information available on selection of appropriate indicators (Kay *et al.*, 1990) the Commission has focused attention on pollutants most likely to cause risk to public health. Annex 1, Table 1 lists the water quality requirements for bathing waters (Appendix IV). The microbiological parameters, which are the most significant in terms of public health and cause for much controversy, have been reviewed considerably. A list of parameters have been deleted including total coliforms, salmonellae, pesticides, metals, nutrients and ammonia. The reasons to leave salmonella from the proposed list of determinands are:

i. the existing limit covers all salmonellae, which vary greatly in their pathogenicity

ii. there is little evidence that illness caused by salmonellae occurs outside of waters that are grossly polluted

iii. salmonellae can enter unpolluted waters by a variety of sources which are not controllable, such as bird droppings (HMSO 1995b).

Member States are required to identify all sources of pollution, and where it is believed that salmonellae might exist remedial measures must be taken and a timetable plan of action be submitted to the Commission.

The enteroviruses parameter, with a standard of zero per 10 litres of water, has been retained. There has been criticism over this standard due to the ubiquitous nature of enteroviruses in the offshore environment making achievement virtually impossible (HMSO 1990b; Grantham 1992). Whilst acknowledging the inherent difficulties complying with this standard, the Commission has held firm on this parameter, arguing that the most important theme for the Directive is to protect public health, some viruses being highly infective. There is potential, however, to replace enteroviruses with a standard for bacteriophages (Section 6.1.2.4), viruses which attack bacteria. Bacteriophages are identified with sewage and have similar resistance to environmental stress and decay rates as viruses (Fewtrell and Jones, 1992; Havelaar, 1993) and therefore could be used as faecal indicators. However, at present there is not sufficient evidence to support numerical proposals and outline procedures required for bacteriophages (HMSO, 1995b). Until this time monthly samples will have to include enteroviruses, unless the Guideline standard for *E.coli* and faecal streptococci have been met in the previous two bathing seasons, and then the frequency can be reduced to two samples per season. In addition, as a consequence of recommendations made by the House of Lords (HMSO, 1995b) a determinand specific to *E.coli* will be included to replace faecal coliforms. Also a Mandatory standard for faecal streptococci of 100 per 100ml with a guideline of 50 per 100 ml has been introduced to replace the previous Guideline criterion of 100 per 100ml.

3.9.3.2 Compliance

Strict adherence to the enterovirus parameter will ultimately lead to increased numbers of beaches failing to meet the Directive. In 1992, data provided by the NRA showed that 21% of the 416 designated waters in England and Wales failed the coliform standards. However, 48% failed the enterovirus standard when most waters were only sampled twice per year and some not at all (ENDS, 1994c). The NRA at the time, expressed concern over the introduction of a Guideline standard for faecal streptococci, which may also affect the number of beaches complying with the new legislation, with only 41% of beaches in England and Wales attaining the current Guideline value in 1993 (ENDS, 1994h).

Compliance with the original Bathing Water Directive (CEC, 1976) caused statistical problems, requiring 95% of samples to meet the microbiological parameters. The UK sample 20 times between the months May to September (Environment Agency, 1997), therefore only two samples not meeting standards means non compliance and failure. Several options have been considered to work around this problem, including relaxation of the 95% compliance requirement and a statistical assessment of all sampling results so that isolated exceedances would not necessarily result in a bathing water being failed (ENDS, 1994c). At present the Commission has chosen to retain the current position on grounds that further complication of the rules would only serve to confuse the public.

Member States have previously had no incentive to aim at Guideline standards set in the existing Bathing Water Directive. The new proposal suggests a category of excellent water quality obtained only by achieving the Guideline standards which should influence tourism. Also if the excellent level is met for two consecutive bathing seasons the sampling may be cut by half.

The proposal does not issue clear rules on non-compliance to deal with breaches, giving Member States a degree of flexibility, but leaves the responsibility in the hands of the individual. The competent authority within each Member State is required to identify the cause or causes of non-compliance and take necessary action to bring about compliance as soon as possible (CEC, 1997). The competent authority must also inform the Commission of the reasons for failure and necessary action to reverse the situation, including a timetable for completion. It would be difficult to enforce legal proceedings on recalcitrant States, but history does show that if necessary Member States risk infringement proceedings for repeatedly failing to comply to EC Directive (Case C-56/90-Rees 1994).

If significant deviation from the Imperative standards occur there is provision made to ban the bathing water on health grounds, taking local conditions into account. Article 7 states that bathing water shall be prohibited '*where pollution constitutes a threat to* *public health'*. In addition to safeguarding bathers provision is also made to ensure information on water quality is prominently displayed, covering whether or not a bathing water complied with the Directive in the previous bathing season, most recent information on water quality and remedial action and timetable for works in progress or planned, Article 5. The Environment Agency and local authorities are responsible for data sampling and information publication at the moment.

3.9.3.3 Impact on the UK

The new Directive to improve the quality of bathing waters has created contention for the UK Government, which was one of the Member States driving for subsidiarity and amendment of the original Bathing Water Directive (ENDS, 1994e). A political gap developed between the Government and the House of Lords Select Committee appointed to consider the original Community proposals (HMSO, 1995b). The Committee contested the Governments attitude that the reforms are unreasonable, and suggested they are not stringent enough. One of the main findings from a report commissioned by the DoE (Kay *et al.,* 1994) was that faecal streptococci concentrations exceeding 32 per 100ml constituted adverse health effects. This is an order of three times lower than the current Mandatory level of 100/100ml. The main issue centred around the faecal streptococci finding which occurred at chest depth in water, 1.3-1.4 m depth, in contrast to the sampling method currently stipulated by the existing Directive, which is 30cm below the surface at 1m depth. Another point of contention has arisen over the methods laid down in the existing and proposed Directives. Havelaar stated that they are mostly out of date (ENDS, 1994a).

With regard to the first set of reforms to the Bathing Water Directive (CEC, 1994), which have not undergone major change in the final proposal (CEC, 1997), it is likely to cost the water companies a further £1 billion to comply with these new standards according to an estimate by the Water Services Association (ENDS, 1994d), in addition to the £2 billion already invested to bring the water quality of bathing waters of England and Wales in alignment with the existing Directive (ENDS, 1994a). The UK Government claimed the initial reforms would provide little if any improvement to health, not warranting the cost involved and that the money could be used more effectively if spent on other areas of health. The House of Lords Select Committee (HMSO, 1995b) has been critical of the Government's attitude to the reforms claiming they have not given enough credence to the findings of the WRc report (Pike, 1994) and that the changes would protect bathers against self-limiting illness. The Committee believed that an acceptable level of faecal streptococci would be 100 per 100 ml, which has now been introduced (CEC, 1997). The Committee also urged the Commission to consider quality assurance programmes for laboratories engaged in analysing bathing water samples and that the standard for enteroviruses should be dropped.

3.9.4 Urban Wastewater Treatment Directive

The EC Directive concerning urban waste water treatment (CEC, 1991) came into existence in 1991 addressing sewage discharge into the aquatic environment (see Appendix IV). Haigh (1995c p.4) reasoned the impetus for the Directive originated 'from a growing concern at the detrimental effects evident in many of the EC's fresh and coastal waters of discharges of inadequately treated sewage'. He further acknowledged the increasing problem to Member States of eutrophication through nitrate and phosphate enrichment in both inland and coastal waters, not accounting for the public health implications. Urban waste water, previously termed municipal waste water includes domestic sewage, industrial waste water and rainwater run-off. The Directive states in the preamble its intention to:

> *'prevent the environment from being adversely affected by the disposal of insufficiently-treated urban waste water'*
> (CEC, 1991 p.1)

The effective result will be to reduce pollution of freshwater, estuarine and coastal waters. It achieves this by setting minimum standards for collection, treatment and discharge of urban waste water including a timetable for achieving compliance. This section of legislation inevitably works in conjunction with the Bathing Water Directive (CEC, 1976) discussed above, in improving the quality of water for recreational purposes and also for reasons outside the interest of this piece of research, such as protection of shell fisheries.

The Directive is comprehensive in defining sewage treatment specifics in terms of population equivalent (p.e.) (NRA, 1991) and detailing dates for reaching the set standards. In general Member States must ensure urban waste water entering collecting systems be subject to secondary treatment (Croall, 1995) before discharge (Article 4). Deadlines to comply vary depending on population size, larger agglomerations of more than 15,000 have until the year 2000; between 10,000 and 15,000 the year 2005 and discharges to fresh water and estuaries with numbers exceeding 2,000 also have until the year 2005.

Provision is made for sensitive waters, criteria for which are defined under annex II. Such waters must receive more stringent treatment (Article 5). Urban waste water discharges from communities between 10,000 and 15,000 p.e. and greater than 2,000 p.e. for estuaries, detailed as less sensitive areas according to Annex II may be subject to treatment less stringent than secondary (Article 6). For particular receiving waters with high dispersion. Such discharges must receive at least primary treatment or been proved to not adversely affect the environment.

Inevitably the economics of accommodating the Directive were heavy and to be incurred over a short time period. OFWAT, with its inclusion of costs to phase out the dumping of sludge at sea estimated a cost of £10 billion (ENDS, 1994j). The DoE initially came up with £2 billion to facilitate changes, and asked the Commission to postpone the deadlines, which it refused to do. A reformed figure of £6 billion has been announce by OFWAT to reach compliance by 2005 (Haigh, 1995). Reservations have been made, notably in 1990 a House of Lords Select Committee (cited Haigh, 1995c) expressed concern over uniform limits, preferring an alternative approach based on environmental quality objectives allowing for natural degradation of sewage in water. Also it was highlighted in ENDS (1994a) that discretionary spending to meet non-statutory RQOs would be cut, therefore areas of conservation not meeting protection from the Directive would suffer, although the DoE has committed expenditure to eliminate this.

The combination of the Bathing Water Directive (CEC, 1997) and Urban Waste Water Treatment Directive (CEC, 1991) have proved a legislative driving force in the improvement of sewage waste disposal leading to cleaner recreational waters. Water service companies are being forced to invest expenditure in implementing new schemes. Further investment will be likely with the implementation of the reforms to the Bathing Water Directive (CEC, 1997). It is too early at this stage to assume the impact of these. However, cautious optimism can be felt in the Principality with Dwr Cymru (Welsh Water) being at the vanguard of sewage treatment schemes in the UK, committing large capital investment (£600m) to coastal schemes to improve it's bathing waters (Western Mail 1995).

Chapter 4

Beach Quality

a)

4.a.1 Problem of Beach Litter

Coastal debris is a serious long term problem, well documented (Dixon and Dixon, 1981; NRA, 1991a; Williams, 1993; Lecke-Mitchell and Mullin, 1997); and any problem associated with litter on beaches is one of universal concern (Willoughby, 1997). Land based pollution takes on many forms from general items discarded by visitors to sewage-related debris and poisonous wastes such as medical objects and chemicals. Effect of this litter creates not only a visual eyesore but can be hazardous to human health (Dixon and Dixon, 1985; Phillip *et al.,* 1997) and harmful to wildlife (Pollard, 1996b). Argardy (1993) echoes Willoughby's view that the problem of litter is international in extent, acknowledging related problems of coastal debris. But he points out that there is a vacuum of literature regarding the aesthetic quality of coastal and marine areas. There is also a dearth of research investigating aesthetics and public perception to coastal water quality (refer Chapter 5) aesthetics in more detail.

Increasing advances in material design, although innovative, highly technical and market driven, are proliferating problems of beach litter and in many cases compounding issues facing coastal zone management (Williams and Nelson, 1997). Certain beaches tend to form perfect sinks for persistent materials which find their way via the marine environment, rivers, estuaries and through visitor carelessness, for example Merthyr Mawr, South Wales. Research on coastal debris is not a new topic (Cundell, 1971). A national survey of litter on beaches was done by Dixon and Hawksley (1980). Since then there has been a progression of work in the UK which has accelerated in volume through the nineties (TBG, 1994b; Mouland, 1994; Galvin, 1996). In particular, a lot of research has been focused on the South Wales coastline (Simmons, 1993; Williams and Simmons, 1996; Williams and Nelson, 1997) Presently there is no definitive answer to curbing debris observed along the UK coastline as sourcing of debris is in its infancy. This chapter reviews origins of litter (Scott, 1972; TBG, 1991a) and corresponding economic effects on tourism (House and Herring, 1995) and management research methods to try and combat the problems of coastal litter (Everard, 1995; Earll and Jowett, 1998). Also beach award systems are reviewed with particular relevance to the area of study and regional initiatives within Wales.

4.a.2 Economic Effects on Tourism

Tourism is a huge industry which is expanding internationally at a high growth rate. With more people travelling to more places than ever before (Steward, 1993) it is not surprising that tourism has become the largest industry in the world (Miller, 1993), and accounts for 66% of all world travel (WHO, 1994b). Statistics produced by the British Tourist Association showed Britain received a revenue of £12 billion pounds during 1996 from 23 million tourist (Quarmby, 1996). The coast encompassing beach and nearshore waters provides an environment conducive to recreation and leisure supporting the biggest tourism trade of any area in the world (Argardy, 1993). The WHO (1994b) also recognise the sea as being the most important environment for tourist movement. In the USA tourism and travel are the biggest national industry, with coastal states receiving 85% of subsequent tourist generated income (CERC, 1996). On a regional scale tourism is also the main industry in Wales on which small communities are financially dependent. The Wales Tourist Board (WTB) estimated that during 1995 Wales received 740,000 overseas visitors who spent *circa* £203 million (Leisure Monitor, 1997).

A day at the seaside potentially presents many hazards such as poor water quality, over exposure to the sun and litter on beaches. Degradation of our inherited coastline will undoubtedly detrimentally affect tourism and the natural environment. It is therefore important from both the demand and supply sides to address the issue of public health, health of the environment and tourism through appropriate management. The health of the consumer is very important (Grant and Jickells, 1995) and can be broken down into two main aspects. First, it is the responsibility of the receiving area to ensure adequate protection of the tourist, such as clean beaches, but it is also the responsibility of the tourist to behave in an appropriate manner to avoid unnecessary risks to themselves, such as using sun screen. Economics is the prime motivating force driving these two factors. However, it is at a higher level of power that the aesthetic quality of the coastline must be protected, through sustainable management planning. House and Herring (1995) found sewage-related debris to have a great impact on enjoyment of natural surroundings, substantiated by the Phillip (1994a) who pointed to the effect of poor aesthetic quality on tourism figures. More recent work by Phillip *et al.,* (1997) also linked environmental degradation from litter and medical waste with loss of income from tourism. The Countryside Commission (1991) has expressed concern over environmental health of the coastline and the social effects on tourism. The government White Paper *The Health of the Nation* (Department of Health, 1992), recognised the need for research to pinpoint the association between health consequences and the quality of the environment.

It has been widely noted that the only real solution to curb littering of beaches is to tackle disposal of waste at source (Simmons and Williams, 1994; Earll *et al.,* 1997). However, this is a very complex task and as an interim measure the only answer is to clean the beaches, although this is curative rather than preventative. Although distribution of litter is site specific depending upon physical aspects and prevailing environmental conditions, a large proportion of litter is not tourist based (Cundell, 1971; Scott, 1972; Simmons and Williams, 1997). Beach clearance is essential to keep them free of debris, although the process is expensive, highlighted by Grant and Jickells (1995). They highlighted high costs borne by the local community. Gilbert (1996) claimed that the indirect cost of clearing the Kent coast of litter in 1995 was £12m. High figures were also estimated for Weston-Super-Mare which cost *circa* £100,000 to clear 2 beaches which attracted 2.5m visitors in 1996, this figure includes both direct costs such as beach raking and indirect costs such as drain blockage (Fanshaw, 1996). These examples are not just restricted to the UK. In Sweden, cost of clearance for the Bohuslan coast for 1994 was in excess of equivalent to £937,000 (Olin *et al.,* 1994). The alternative can be more devastating though. Statistics quoted by Fanshaw (1996) showed that closure of beaches in New York/New Jersey due to pollution cost *circa* £2 billion over one season.

Tourism is the fastest growing industry in the world. To compete in this highly competitive dynamic market, beach quality must be maintained and improved to attract tourists (CERC, 1996). Growth of tourism and the economic boom that comes with it is not mutually exclusive from human and ecological impacts. Miller (1993) reported that marine tourism in particular has become inherently controversial and stated that resolution of these problems lie in:

- Scientific analysis of environmental and social conditions
- Policy analyses
- Planning
- Public education

These sentiments were similarly expressed by Steward (1993) who also noted positive and negative effects of tourism on natural environments. His opinion on the associated problems was to include community participation in planning if successful management was to be achievable.

4.a.3 Litter Types and Origins

All landscapes are often degraded due to presence of debris. Beaches tend to provide the destination for both anthropogenic input of pollution and sinks for natural debris, such as drift wood, algal blooms and seaweed. Although the main emphasis of this report concentrates on input of waste due to man the latter also has serious implications for beach management. Distribution of algae along the shore can be poisonous to humans and decaying organic matter produces offensive odours which affect the enjoyment of a visit to the coast (Green and Birchmore, 1993).

Extrapolating litter back to its absolute origin, for example production, is difficult, but it is possible to analyse the pathways used for debris to reach shorelines. In the context of this discussion the term source does not refer back to the manufacturer but to the pathway, for example a river. There is debate over the origin of litter. The TBG (1997b) linked litter found on the Cumbria coast to originating from an international sources. Data recorded by the Norwich Union Coastwatch project also identified coastal litter on the British coast stemming from 27 other countries (Rees and Pond, 1994). There are three main sources of marine litter:

- Visitor discards
- Marine debris, from dumping waste overboard
- Estuarine/riverine input including combined sewer overflows
- Sewerage system outflows

The Third International conference on Marine Debris in 1994 (Faris and Hart, 1995) claimed that land-borne sources account for at least 70% of coastal litter. It is more likely that the litter source is dependent upon other physical and environmental variables such as geographical position, geology, aspect of beach, prevailing winds and proximity to urbanisation. Simmons and Williams (1997) claimed that 80% of litter deposits on South Wales estuarine beaches is riverine in origin.

It has been proven that in many instances, tourist input to beach litter is not the prime contributory factor in coastal pollution (Simmons and Williams, 1992). Further work by Williams (1996) showed no seasonal change in litter quantities along the Glamorgan Heritage Coast. Earlier findings by Scott (1972) also showed that on remote Scottish beaches there was a build up of litter regardless of limited tourism in the area. Marine borne debris is a serious concern and is the most difficult to deal with. Williams and Simmons (1995) listed the main sources of ocean debris:

- Merchant shipping
- Military shipping
- Commercial fishing
- Cruise ships
- Recreational vessels

The MCS have highlighted the extent to which ocean dumping affects the marine environment(Taylor, 1996). Ships are reported to dump at least 4.8 million pieces of metal, 450 000 plastic items and 300,000 containers into the sea every day (Taylor, 1996). This adds to the magnitude of marine debris in offshore waters, of which the public are unaware (Lecke-Mitchell and Mullin, 1997). Work done in Swansea Bay also found significant proportions of debris on the sea bed (Simmons *et al.,* 1993).

The final source of beach litter is from the sewerage system, industrial and domestic. Developments in material design, especially plastics have outgrown the capacity of the sewerage systems, which are not designed to cater for modern consumer items (Lowe, 1996). One of the predominant findings of the Norwich Union Coastwatch (1996) project found an abundance of sewage-related debris on the British coast, on average 32 items per mile. In addition it also found an average of 2 items of medical waste per mile along the British coast (Rees and Pond, 1994). Presently only half of Wales sewage is treated (Lowe, 1996) and parts of the sewerage system are now well outdated, being built in Victorian times (Welsh Water, 1996a). This adds pressure to South Wales which has received the resultant waste produced by the heavy industrialisation of the local Valleys from the mining and steel industries which coupled with the geography of the area with major south flowing rivers has over time contributed to polluted beaches and shorelines of the area. Even though new schemes dealing with sewage are underway in Wales (Mason, 1995a), a pressing problem is the numerous number of combined sewer outfalls, the majority of which are unscreened (Williams and Simmons, 1995). Combined sewer outfalls are integral parts of the sewerage system and act as relief valves which operate under conditions of heavy rainfall and discharge into the riverine system. It is estimated that there are 2500 in Wales (Welsh Water, 1996a).

Coastal litter can be broken down into five main categories (Dixon and Dixon 1985; Earll 1997a, 1997b; Phillip *et al.,* 1997; Williams and Nelson, 1997):

1. General
2. Sewage-related debris
3. Hazardous
4. Medical
5. Accumulations

General litter is a cocktail of items which are often left by tourists and visitors to the coast or washed up and do not fall into one of categories 2-4. They comprise such items as aluminium cans, confectionery wrappings, polystyrene containers, soft drink bottles and paper for example. Although debris found on beaches varies widely, 20 items make up for 75% of all litter (Pollard, 1996a). This figure is narrowed further with 75% of rubbish found on the Glamorgan Heritage Coast to be plastic (Simmons and Williams, 1992). Plastics are frequently mentioned in the literature as being the most abundant material found on coastlines (Pruter, 1987; Green and Birchmore 1993). Polythene, polystyrene, PVC and polypropylene comprise the majority of plastics found on the coast. Willoughby (1996) also found polystyrene and plastic to be the most voluminous on beaches in agreement with Williams and Nelson (1997). Green and Birchmore (1993) pinpoint the main reasons large numbers of plastics are evident on the coastline and why:

- Lightweight and high mobility
- Strong
- Decay resistant
- Low density

Demonstrating the resistance of plastics to environmental stress, the TBG found a plastic bottle on the Scottish coast which was 20 years old (TBG,1991a) and more recently (1998) a bottle was found at Merthyr Mawr, South Wales, that was 31 years old (Williams *pers.comm.,* 1998). Green and Birchmore (1993) make clear the necessity to remove plastics from water courses and beaches to avoid major environment problems. The MCS claim that more than 1 million birds and 100,000 marine mammals and sea turtles die every year as a result of eating or getting tangled up in plastic (Taylor, 1996). The composition of sewage-related debris is mainly feminine hygiene items, condoms and nappies. Simmons and Williams (1994) point to the problem of dealing with these objects, which have similar characteristics to those outlined above by Green and Birchmore (1993) due to a large proportion being plastic or containing a plastic component. Simmons and Williams (1994) noted their multiplicity of inputs, mobility on the beach and longevity of life and slow breakdown to be the most difficult facets to deal with. These items have a strong impact on visual appearance of the coastline (House and Herring, 1995) and potential to affect economies dependent on beaches. Hazardous debris and medical wastes which are becoming more wide spread, include hypodermic syringes and needles (Green and Birchmore, 1993). These items are harmful to beach users and detract from the aesthetic coastline which could also be detrimental to tourism revenue (Phillip *et al.,* 1997). The last category relates to accumulations of litter which are often windblown and form unsightly combinations of all litter types. Recent research has addressed accumulations of litter as a separate category from generic types of debris such as plastic (Earll, 1997a). Quite often these are found above the high water mark.

The 6th Clean World International Conference held in Paris 1978 identified widespread environmental impacts of marine litter (cited TBG,1991a). These included aesthetic appearance of beaches, danger to living organisms, affect on coastal amenities and pleasure craft and ships. Already as mentioned the impact of this litter is having dire consequences on wildlife. The aesthetic value of the coastline affects the way in which the public perceive their immediate surroundings, affecting enjoyment of tourists and recreationalist. The public are becoming increasingly aware of pollution, although there is a vacuum of literature addressing this issue (Williams and Nelson, 1997).

4.a.4 Responsibility and Policy

Problems created by coastal pollution have been highlighted from the perspectives of health, conservation and aesthetics. Coastal water quality is governed by the EC Bathing Water Directive (CEC, 1976). However, there is no specific law which relates explicitly to beach litter. Marine borne debris is governed by the MARPOL convention (MARPOL, 1973/1978). Fifty percent of the worlds shipping countries are signed up to the convention which addresses control of discharge of pollutants at sea. Although in theory this convention is worthwhile its pragmatic application is ineffective due to limited ability to enforce and police on the high seas.

The main UK legislation pertaining to sources of litter which potentially find route to beaches are the Environment Act (HMSO, 1995a) and the Environmental Protection Act (HMSO, 1990a). The Code of Practice on Litter and Refuse, Section 89 of the Environmental Protection Act requires local authorities to ensure their land is kept clean and free of litter and refuse. This includes beaches under their jurisdiction. The Environment Act (HMSO, 1995a) also gives local authorities power to inspect an area to detect statutory nuisances and serve an abatement notice on those responsible. This covers any sewage pollution created by water authorities. In addition European legislation affecting the origin of marine litter is the European Council Directive on waste (CEC, 1975), amended in 1995 (CEC, 1995) concerned with controlled waste which includes household, industrial and commercial. It is the responsibility of the Environment Agency to enforce these laws.

4.a.5 Measuring and Monitoring

Beaches vary considerably in their size and geography, existing under a wide range of dynamic environmental processes. These factors influence the build up of marine debris, its composition and spatial distribution. In order to effectively manage and control land-based coastal pollution it is essential to be able to objectively measure it. At present there is no standardised procedure to deal with the complex task of measuring marine litter

(Williams and Simmons, 1997): although there are a number of techniques which have been designed for specific purposes by particular groups. A brief outline of these are described below.

4.a.5.1 Garber Index

The Garber Index (1960) is a fairly crude survey developed to measure aesthetic quality of beaches. The Index uses a log sheet (Appendix V) sectioned in two parts for the sampler to record results. Section A is related to the condition of the water including colour, mineral oil, surface active substances, phenol and tar/floating materials. Each attribute for section A has a binary response to represent either presence or absence. Section B relates to the beach area broken down into strandline, inter-tidal region and waters edge. The sampler is required to record material present on the beach, including intact faeces, grease/scum, sewage debris, contraceptives/ tampon applicators, sanitary towels and noxious sewage odours. These materials are graded on a four point scale: 0- absence; 1- trace; 2- some material at intervals and 3- sufficient to be objectionable. In addition the sampler is required to record prevalent environment conditions such as weather, wind, state of the tide, turbulence of the sea and anthropogenic activities, including bathing and beach users. The technique has been widely employed in aesthetic quality assessments and use in the past by the NRA for monitoring estuarine and coastal sites (Everard, 1995).

4.a.5.2 Environment Agency Pollution Incident Categories

The Environment Agency have designed a system to measure litter using a 4 category system (NRA, 1995b). Each category is classed into four grades:

i. Sewage-related debris
ii. General litter
iii.Harmful litter
iv.Accumulations of litter

These categories are graded on a four point scale in increasing severity of incident from A to D, with respect to 100m stretches of beach. Each category is weighted such that only very few Harmful items (1-3) are required to reach a C grade whereas for General litter, which is not perceived to be as offensive, requires up to 10 items to reach grade C. This system, although fairly basic is practical and understandable allowing comparisons to be made and hot spots to be identified.

4.a.5.3 Thamesclean Project

The Thamesclean Project (Lloyd, 1996) four grade A-D system was used in the Code of Practice on Litter and Refuse developed for the 1990 Environmental Protection Act (cited Earll *et al.,* 1997). The method uses a similar system of grading to the Environment Agency. Four grades are established which distinguish between quantities of litter:

Grade A:	Absent: no evidence of litter anywhere
Grade B:	Trace: predominantly free from litter apart from a few small items
Grade C:	Unacceptable: some at intervals; widespread distribution of litter with minor accumulations
Grade D:	Objectionable amount: area heavily littered, with much accumulation

The Thamesclean Project is operational. However, it is subjective, does not giving any indication to the extent to which a grade functions, does not allow direct comparison of beaches and is also open to surveyor bias.

4.a.5.4 NRA Aesthetic Survey of Beaches in the South West

A two year survey investigating the aesthetic impact of crude sewage discharges on popular beaches in the South West was conducted by the NRA over 1990 and 1991 (NRA, 1991a). A modified Garber Index was used and surveys were conducted along the high water mark and along the water's edge. The method attempted to score beaches on a scale with increasing quantities of sewage items from 0 to greater than 9. Four categories A - D were set up:

A Free from sewage-related debris:	0
B With trace quantities of sewage-related debris:	>0 - 1
C With intermittent quantities of sewage-related debris:	>1 to <9
D With objectionable quantities of sewage-related debris:	>9

The sampling area included a 10m transects straddling the sample lines. Beach units were created at 100m sections for beaches <500m in length; 200m sections for beaches 500m - 1km and 500m sections for beaches >1km. All sections were continuous. Measurement of grease/scum and noxious sewage odour introduced a degree of subjectivity.

4.a.5.5 NRA Aesthetic Assessment and Management

The NRA have designed a system of General Quality Assessment (GQA) to grade water quality for rivers and estuaries and coastal waters (Everard, 1995). These are split into discrete windows. Parameters to grade water quality for estuaries and coastal water include general chemistry, nutrients, biology, sediment quality and aesthetics. Although specific to water quality the aesthetics window accounts for public perception and runs parallel with the more conventional parameters described above and considers land-based marine debris. Transects 40m wide are surveyed along the strandline, inter-tidal zone and paddle zone (items visible within 10m when standing knee-deep in water). Objects are recorded dependent upon type. For estuaries and coastal waters general litter are counted as items with a dimension less than 30 cm, such as drink cans; gross litter are items with a dimension greater than 30 cm, for example car tyres, and sewage-related debris includes for example feminine hygiene items and condoms. Four other categories exist oil, foam and scum which are measured according to surface area, unattached seaweed; faecal matter of non-human origin and colour. A weighting system is being developed to derive an overall aesthetic score dependent on public perception (Everard, 1995).

4.a.5.6 Norwich Union Coastwatch

In 1989, the Norwich Union Coastwatch project was instigated as part of a European initiative to measure and monitor coastal litter (Rees and Pond, 1996). The national scheme is based at Farnborough College of Technology and uses groups of volunteers to walk pre-defined stretches coast. The total coast measured is estimated at approximately 1145 miles. Volunteers are required to record litter and other items such as biota and sewage inflows. Litter categories are classed as gross, moderate and slight. Sewage-related debris and medical waste are considered to be the main indicators of beach aesthetic quality and visitor health risk. The project is now in its 8th year and has built a database on pollution of beaches (Appendix V).

4.a.5.7 Marine Conservation Beachwatch

The MCS have a programme called 'Beachwatch' which addresses marine litter. The project is run annually and differs from the other methods mentioned in that it attempts to clear beaches by recruiting volunteers through local campaigning strategies as well as recording the litter (Pollard, 1996b).

4.a.6 Standardising Methods

The methods and projects defined are not a comprehensive list but are of the more prominent national schemes and methods for measuring and monitoring marine litter in the UK. Other studies have been carried out such as recording of individual items of litter along the South Wales coast (Williams and Nelson, 1997). The systems mentioned are designed to fulfil particular aims and the respective bodies have their own agendas, which are not compatible with each other. It is now recognised that to improve the coastline it is necessary to create a standardised approach so that different methods can be harmonised and data pooled to build a more complete national picture of marine debris on British beaches (TBG, 1991a). Earll (1996) recognised this need and set out criteria that a standardised method should meet.

- robust and effective
- repeatable and could be used routinely
- quick and cheap to undertake
- linked to management and prevention
- enable the status of beaches to be reported in easily understood terms: 'dirty' 'clean' 'very clean'
- widely recognised by national agencies and local authorities

4.a.6.1 The National Aquatic Litter Group

The National Aquatic Litter Group (NALG) (Earll and Jowett, 1997) has evolved from a series of meetings and consists of a wide range of sectors including the Environment Agency local authorities, academics, TBG, water industry and non-governmental organisations (NGOs). The aim of the group is to create an effective management protocol for the prevention of litter. A key element was to create a standardised compatible approach to compare results of different existing surveys, to assess beach litter on a national basis (Appendix V). The protocol was derived primarily from 2 models, the Environment Agency Pollution Incident Categories and the Thamesclean Project. This is the probably the most progressive attempt at producing a standardised technique for measuring and monitoring beach litter and has the support from a broad spectrum of influential bodies. The protocol (draft 6) is in operational format, and is due to be endorsed in 1998 by a meeting of the engaged bodies involved. Earll and Jowett (1997) give a detailed review of the methodology. A summary of the approach is described below.

The main principles are to describe a beach on an A-D basis from Very Clean, Clean, Dirty and Very Dirty. Three zones are analysed, above high water, the strandline zone and the inter-tidal zone. Seven categories are delineated:

- sewage-related debris (e.g. feminine hygiene items and condoms)
- gross litter (items with a dimension greater than 50 cm, e.g. shopping trolley)
- general litter
- harmful litter (items deemed to be dangerous to human health or animals, e.g. sharp glass and used syringes)
- accumulations (items with a dimension less that 50 cm, e.g. coke cans, cigarette packets)
- oil (based on absence or presence and whether it is objectionable)
- faeces (the numbers found in each zone, usually dogs)

Surveyors are required to walk in a zig zag pattern between zones with a maximum span of 50 m and record the litter using the category sheets. Data is collected to the back of the beach, above spring high tide, be it a seawall, dune etc. Each category is assessed individually and attributed a grade from A-D depending on quantity. The grades vary depending on the category. For example a grade D for general litter is more than 1000+ items, whereas a grade D for harmful litter is only 4+ items. Once all the categories for a particular beach have been graded, the process goes into phase 2 where each category is then weighted depending on perceived offensiveness. Finally, the numerically weighted categories are summed up and the score compared with a table which classifies the beach as Very Clean, Clean, Dirty or Very Dirty.

This system is operational and can be employed by surveyors with little training. Final weighting of the beach makes it comparable allowing different agencies to utilise the system creating a database of marine litter on British beaches allowing identification of hot spots. Recent work (April, 1998) by Williams (*pers.comm.)* has shown that the educated lay person can utilise this scheme, which has proved to be a robust technique, easy to use and no difficulties existed between different groups assessing litter along the same transects. However, consideration must be given to the weighting system as the process of weighting inherently involves the loss of raw data.

Earll (1997b) has suggested that further work should look at categorising litter into coastal species and identifying not only the material but also the function to identify with a source. For example observation of paper gives little knowledge of its origin. However, if this information was coupled with its function, such as sweet wrapping it could be inferred that there is a strong possibility that its origin is from a younger social group. Similarly the principle can be applied to cigarette packets.

4.a.7 Beach Award Systems

Problems related to marine debris have been discussed and practices to deal with sustainable management of the coast. It is apparent that beach quality is directly associated with recreational enjoyment and tourism economics. Several seaside initiatives have been established to bridge the gap between management and beach marketing. There is much controversy over the numerous different schemes available and whether they are actually beneficial to the public or whether they just serve to confuse: for example the European Blue Flag, Golden Starfish Award, MCS Dolphins, TBG Seaside Awards, Resort and Rural (TBG, 1991b; TBG, 1994b; Stanton, 1997; FEEE, 1997). In general they provide information on the beach environment, safety and facilities. The different systems in operation are analysed below and their implications to Wales.

The seaside award schemes discussed are aimed at identified beaches. Initially beach designation criteria came under the code of practice of the EC Bathing Water Directive (CEC, 1976). The Directive stated that 500 people had to be in the water at any one time. This excluded most beaches in Britain including Brighton and Blackpool. The first count of designated beaches in the UK was 23. However, there are now new criteria which identify beaches, totalling 472 (TBG, 1997c).

4.a.7.1 European Blue Flag

The European Blue Flag is probably the most renowned of the seaside award flags. Although the scheme was conceived in France, 1985, it was introduced formally to Member States during the European Year of the Environment by the Foundation for Environmental Education in Europe in co-operation with the Commission for the European Communities (FEEE, 1987). The FEEE is a network of organisations set up to promote environmental education in Europe. The framework through which the Blue Flag Campaign is run recognises the concepts of Local Agenda 21 (960UN, 1992) and involves integration of concerns, co-operation and partnerships at local, regional and national level (FEEE, 1997). These are key elements outlined in Agenda 21, Chapter 17 of which addresses marine and coastal environments (UN, 1992). The award is aimed at resort beaches and marinas. A resort beach is defined by FEEE as one which:

> *'actively encourages visitors, has developed its facilities and provides varied recreational opportunities. The beach must be adjacent or within easy and reasonable access to the urban community and typically would include all the following facilities: a café or restaurant, shop, toilets, public transport, supervision, first aid, public telephone'.*
>
> (TBG, 1996a p.3)

An independent body is set up in each country which is responsible for allocation the Blue Flags and monitoring of coastal resorts. In the UK the TBG are the governing agency (Stanton *pers.comm.*, 1996). The Blue Flag is only valid for one year, and beaches are assessed 3 during a bathing season to ensure compliance. Failure to maintain standards will result in the Flag being withdrawn. The water quality criteria uses a retrospective approach being based on the previous years results (FEEE, 1997).

The Blue Flag is presented to beaches that match 25 strict criteria divided into three groups, Environmental Education and Information, Environmental Management, Water Quality and Safety and Services (FEEE, 1997). Initially quality of water at beaches was required to meet with Mandatory standards set by the European Bathing Water Directive (CEC, 1976). These centre around bacterial density for total coliforms and *E.coli*, (Table 4.1). Revisions to the Blue Flag came in 1992 which saw the criteria for water quality become 20 times more stringent requiring the Guideline standards set in the Bathing Water Directive. These included the faecal streptococci parameter (Table 4.1). Yet the derivation of these standards is questionable having no sound scientific evidence to support their relationship to health (Cartwright, 1992 Kay *et al.*, 1994). Rees (Robens Institute, 1997) is also cynical about the use of these standards in setting criteria for seaside awards stating that they do not guarantee quality of bathing waters and suggested that public health issues should be divorced from amenity issues.

The introduction of new standards required in 1993 saw nearly a 50% drop in UK beaches failing to attain the Blue Flag, dropping from 35 to 20 (TBG, 1994b). Eighteen parameters are set for beach management and safety which range from being free of industrial and sewage discharges in the beach area to provision of frequently serviced litter bins and lifeguard cover. The third category deals with education and information covering 6 criteria including laws on beach use and code and posting updated information on bathing water quality (FEEE, 1997). A full set of parameters laid out for the Blue Flag can be viewed in Appendix VI.

Out of the 472 identified beaches in the UK, 192 qualified with Guideline water quality standards (CEC, 1976) but only 38 complied with the full range of parameters laid down for attainment of the Blue Flags (TBG, 1997c). The number of European Flags awarded to UK beaches has not altered significantly since 1991 when 35 beaches were successful. However, the water quality criteria has changed, which implies that the quality of UK beaches are improving. More locally nine of 1997 Blue Flags were obtained in Wales, constituting nearly 25% of those achieved for the whole of the UK. This number has over doubled from the 4 attained in 1996 (FEEE, 1996).

Parameter	Imperative std. (I)	Guideline std. (G)	Compliance Rate %
Total coliforms	<500	<10 000	80
E.coli	<100	<2 000	80
Faecal streptococci	<100	--	90

Table 4.1 EC Bathing Water Quality Parameters (values per 100ml)

4.a.7.2 Tidy Britain Seaside Awards

The Tidy Britain Group (TBG) are recognised as the national litter abatement agency working closely with communities, central and local government for improved local environments (TBG, 1997). Part funded by the UK Government (£2.5m) they are an independent agency and registered charity (TBG, 1994b) responsible for allocation of the main seaside award schemes in the country.

In 1992 the TBG brought out their own Seaside Awards (TBG, 1992). The two new Seaside Awards encompassed both resort beaches located in or near towns and rural beaches found in remote places with limited facilities and supervision. The resort beaches required water quality which had met the Mandatory standards and also met 28 land-based criteria covering Beach and Inter-tidal Area, Safety, Management, Cleansing provision and Information and Education. The Seaside Award (resort) could be further upgraded to the Seaside Premier Resort if the water quality reached Guideline standards. Rural beaches were also required to meet Mandatory water quality standards in addition to 12 land-based criteria, under the same headings as the resort award, but less detailed. Similar to the resort award, the rural category could be upgraded to a Premier Rural award by reaching Guideline standards (TBG, 1992). The judging of the beaches were initially on a one off basis, but since 1996 have been increased to three inspections during the bathing season. In 1994 there were 54 Welsh entries for the Seaside Award Schemes. There were 16 Premier Seaside Awards presented, 32 Seaside Awards and 6 fails (TBG, 1997). The criteria for the Seaside Awards changed following the 1994 season, discussed below. It is therefore not possible to make a direct comparison between the number or awards obtained in 1996 to 1994. However, there were 65 successful Seaside Awards applications in Wales for 1996 (TBG, 1996b).

The introduction of these schemes came under much disputation, being coincident with the timing of the European Blue Flag upping its water quality requirement (TBG, 1992). The implication of the changed standards for the prestigious Blue Flag was grim for UK beaches most of which were striving to barely qualify for the much less rigorous Imperative standards. Environmental groups believed the Seaside Awards were a ruse to get around the more stringent Guideline water quality parameter for Blue Flags. The MCS claimed in The Guardian (1994) that they were the result of political pressure to add to the confusion at the seaside, backed by the editor of the Good Beach Guide who said they were just an attempt for tourism bosses to fly a flag meeting minimum standards (The Guardian, 1994). The TBG, English Tourist Board and British Resorts Association countered by pointing out that the Seaside Awards catered for rural beaches, which the Blue Flag fails to do and that their role was to promote beaches irrespective of water quality (Guardian, 1994). In support of the TBG awards 91% of Seaside Awards obtained in Wales for the 1996 season were rural.

In 1995 the TBG Seaside Awards were reviewed and changed the Premiership grade was perceived to be superfluous with respect to the Blue Flag. There are now two Seaside Awards, resort and rural. They have retained their land-based criteria but require only the mandatory water quality standard. The fee for the TBG Seaside Award flags is £400 with an additional £200 required for consideration as a Blue Flag beach. The fee includes the flag/plaque, administration, judging and resort information poster costs (TBG, 1997). Table 4.2 compares criteria for the Blue Flag and Seaside Awards.

	Water Quality Criteria	No. of Land-based Criteria
Blue Flag	Guideline	24
Seaside Award Resort	Mandatory	28
Seaside Award Rural	Mandatory	12

Table 4.2 Criteria for TBG Seaside Awards and the EC Blue Flag

The TBG also implemented a Golden Starfish Award (TBG, 1991b) in co-operation with Greece as a pilot project in 1991 (Stanton *pers.comm.*, 1997). The Award was intended for European beaches not eligible for the European Blue Flag due to their smallness in size and rural nature. The Award required high water quality standards and spotlessly clean beaches. Other criteria included ease of access and protection of a warden or guardian to ensure that the beach quality were maintained (TBG, 1991a). The Golden Starfish Award was only operational for one year, during 1991.

The three current flag systems, the Blue Flag and Seaside Awards are designed to encourage high beach quality and motivate local authorities to attain the required standards to aid in promoting their beach. The TBG have also produced a code for beach users to encourage them to enjoy the beach whilst conserving the environment. These include advice on disposal of litter and safety (TBG, 1991a).

4.a.7.3 The Marine Conservation Society Good Beach Guide

The Marine Conservation Society (MCS) are an environmental organisation and registered charity set up to safeguard the marine environment across a whole range of conservation issues (MCS, 1996). In partnership with the Readers Digest they produce the annual Good Beach Guide which gives information on the quality of UK beaches.

Each beach is listed with a range of attributes which include information on flags, water quality, cleanliness, access, toilets, food and conservational detail. The main criterion is the water quality standard which is graded via a 5 point scale, represented by a dolphin symbol (Table 4.3). For recommendation the beach must achieve at least 3 dolphin status (MCS, 1997a).

Grade	EC Bathing Water Standards: Mandatory (I); Guideline (G)
fail	less that 95% EC Mandatory standards
one dolphin	95% pass of EC Mandatory standards
two dolphins	100% pass of EC Mandatory standards
three dolphins	100% pass of EC Mandatory standards and 80% pass of Guideline standards Coliform standards
four dolphins	100% pass of EC Mandatory standards, 80% pass of Guideline standards Coliform standards and 90% pass of Guideline faecal streptococci standards

Table 4.3 Good Beach Guide Classification

The MCS also produce a Seashore Guide in conjunction with the Countryside Council for Wales (MCS, 1997). The aim of the guide is to protect the beach environment giving instruction to beach users on how to act responsibly regarding rubbish and living organisms in conserving the coastline.

4.a.7.4 Beach Rating Schemes

The beach award systems mentioned above do not take into account the perception of the beach user (Morgan *et al.,* 1993). An innovative check list was devised to rate beaches in the south west Peninsula on physical, biological and human usage parameters (Williams *et al.,* 1993). The attributes were scored on a five point scale and totalled to give a percentage score for each beach. Over 180 beaches were surveyed and the results tended to follow European Blue Flag beaches. Further work has been done on beach ratings in the Coastal Research Unit at University of Glamorgan. The rating scheme took account of beach user perception to enable the public and coastal managers to compare beaches (Williams and Morgan, 1995). The system produced has been used to survey 70 beaches in Wales and further surveys have been undertaken on the south coast of England, Mexico, California, Catalonia (Spain) and Turkey. The research attempted to objectively quantify a beach by attributing a numeric score. Morgan's (1996) results from using this beach rating scheme showed priorities were good bathing water quality, presence of clean toilets, banning of dogs and absence of oil contamination, sewage debris and litter. The results also found that higher social class placed more emphasis on pollution, but no relationship between economic class and beach preference was evident. Morgan (1996) noted the importance of environmental quality for attractiveness and its affect on tourism. This work is the first attempt at investigating human parameters in beach rating schemes.

4.a.7.5 Comparison with Europe

A report by Which Magazine (1994) noted that Britain's beaches came second from bottom in complying with the European Bathing Water Mandatory standards, with only Germany producing worse results. However, it commented that results given by other Member States are not necessarily reliable. Half the countries - France, Germany, Greece, Italy, the Netherlands and Portugal failed to test their beaches often enough to make the results valid. At least two countries (Italy and Greece) admitted in the EU's official report that they have ignored results of samples taken after rainfall when readings are likely to produce lower results. Inspectors commissioned by Holiday Which, tested five beaches in Italy that had claimed to meet the EC Mandatory water quality standards. All five beaches failed to meet the Mandatory standards (Which Magazine, 1994). There is obviously a need to standardised sampling procedures (section).

4.a.8 Initiatives in Wales

4.a.8.1 The Green Sea Initiative

Almost 75% of the Welsh shoreline is designated for its natural beauty, amenity value or scientific interest, the economy being heavily dependent upon coastal tourism (Welsh Water, 1997). The Green Sea Initiative has been formed to develop Welsh beaches to produce both social and economic benefits (WTB, 1997a) maximising Welsh Waters £600m capital investment into sewage treatment over a planned five year period between 1995-2000 (WTB, 1997b). The Green Sea project is a partnership between a diverse range of bodies including local authorities, statutory agencies, the private sector, environment organisations and voluntary organisations (WTB, 1997). Launched in May 1996 and named after a poem by Dylan Thomas, it has been described by Welsh Water as:

> *'The aim of Green Sea is to make the coastline of Wales the pride of Europe. (It) is a unique joint venture involving more that 30 public and private organisations which are concerned with the environment...Green Sea will protect a national asset of incalculable value, ensuring the highest environmental standards around the coast'*
> (Welsh Water, 1996a p.20)

The key goal of Green Sea is achievement of 50 European Blue Flags across Wales by the Millenium (FEEE, 1997; WTB, 1997b). Selection of the Blue Flag above other existing awards is the requirement of the Guideline water quality standards set by the EC Bathing Water Directive (CEC, 1976). As already mentioned these are 20 times more strict than the Mandatory standards, stipulated for seaside flag systems such as the TBG Seaside Awards. The objective is not just to obtain Blue Flags but to raise water quality across 80 other beaches too remote to receive the prestigious European status (TBG, 1997a). Presently there are nine Blue Flags in Wales, an increase of over double since last year (Welsh Water, 1997). To keep on target a further 17 have been applied for next year (WTB, 1997a).

Green Sea is chaired by the WTB but has a network of local groups set up to act as beach guardians. The emphasis on public involvement at local level, including local authorities gels well with objectives laid down in Agenda 21 on sustainable development (UN, 1992). The equivalent cost for achieving the desired high water quality standards across beaches in Wales will be £500 per household. In 1995 research conducted showed 61% of people in Wales were not satisfied with the quality of sea water in our bathing areas and 96% think it important that this is improved (Welsh Water, 1996a). To encourage beaches to reach the Guideline water quality parameter the WTB has made £200, 000 available for local authorities to clean up beaches in order to obtain the Blue Flag (Maguire, 1997). However, this does not take into consideration the apparent low level of understanding the public have in relation to beach award systems, including the Blue Flag (House and Herring, 1995; Nelson and Williams, 1997a; Nelson *et al.,* in press).

4.a.8.2 Coastal Forum in Wales

A governmental initiative set up the 'Coastal Forum in Wales' in March 1997 (Welsh Office, 1997). The purpose of the forum is to provide a platform for communication, resolution of conflicts and promotion of coastal interests in Wales. Similar to the Green Sea Initiative a diverse spectrum of interests are involved including academics, Welsh Water, WTB, Keep Wales Tidy and local authorities. Two meetings have been convened, but as yet little information on developments of the forum have appeared in the literature.

4.a.8.3 Severn Estuary Strategy

The three beaches studied in this research lie on the Bristol Channel, Whitmore Bay lying within the boundaries of the dynamic Severn Estuary. The Severn Estuary Strategy has been developed by local authorities, Government agencies and other organisations to co-ordinate and integrate effective management of activities and developments along the South Wales coastline affecting the estuary. These organisations include ports and harbours, business and industry, conservation, recreation and archaeological groups. The aims are to promote sustainable use, resolve conflict of use and promote strategic planning of estuaries (Severn Estuary Strategy, 1997). A joint Issues Report has been compiled (Severn Estuary Strategy, 1997) which addresses key issues along the estuary by forming working groups to form the Strategy partnership. It is the intention of the Severn Estuary Strategy (1997) to accommodate human activities along the estuary, which are extensive, whilst at the same time safeguarding the natural environment for future generations. To achieve this the Strategy provides a platform for the communication and co-ordination of all those whose actions effect the estuary.

b) *Perception of Coastal Pollution*

4.b.1 Literature Limitations

Aesthetic quality of the coast is infrequently addressed by studies based on aquatic environments. There is also a dearth of literature concerned with the way in which the public perceive natural sea and landscapes which have been visually impaired by pollution. The terms aesthetic and perception are explained here to contextualise the following review. 'Aesthetic' is defined as 'concerned with beauty or appreciation of beauty' and 'perception' is defined as 'the ability of the mind to refer sensory information to an external object as its cause' also 'the intuitive recognition of a truth, aesthetic quality etc..' (The Oxford Dictionary, 1991). The second definition intrinsically links the two terms, so one cannot be considered fully in isolation from the other. To comprehend perception it is necessary to understand the brain's function to interpret its surroundings or immediate environment through the sensory system involving processed information of touch, smell, taste, hearing and sight. Preceding water research has highlighted sight as being the most prominent of the senses in determining beach and coastline aesthetics (David, 1971; Smith *et al.,* 1995b).

Attitudes to the natural environment, in particular the coast are changing rapidly. A study conducted by the Department of Health illuminated this fact showing public concern over sewage contaminated beaches and bathing waters was ranked second to chemical pollution, reported by The Times (1991). A greater public awareness is being expressed through recognition of the sensitivity and vulnerability of their surroundings. The coastal zone is coming under increasing pressure from development and over use, acknowledged by Borrego (1996 p.4) who stated '..rich natural resources of coastal regions, like those of coastal waters, are under very strong pressure. The threats to the coastal zone are enormous.' Water related recreation for example has markedly increased over the past two decades, adding pressure and competition for this prime resource space (WHO, 1994b). Borrego (1996) specifically identifies the strain on coastal waters, which play an important role for recreation. However, research suggests that poor water conditions create a risk to health (Cabelli, *et al.,* 1982; Balarajan *et al.,* 1991). It is also important to be aware that irrespective of the quality of water, its appearance *per se* is highly influential in peoples perception of the coast (David, 1971). Sound management principles must be employed reliant upon nurturing sustainable use of resource whilst encouraging enjoyment of the coastal zone. Therefore it is not just the physical health of the beach user that is important but also the psychological health of the beach user in coastal management issues.

A substantive volume of literature exists regarding coastal water quality standards (Kay *et al.,* 1990; Wheeler, 1990; Phillip, 1994b) and health risks (Cabelli, 1983; Rees, 1993), (refer Section 3.3). However, little research has investigated perception of marine and beach pollution, acknowledged by Young *et al.* (1996) or aesthetic quality of the coastline (Morgan *et al.,* 1993; Williams and Morgan, 1995; Williams and Nelson, 1997). Studies on perception of water quality is a relatively new topic, evolving over the past 25 years, starting in the early 1970s (David, 1971; Nicolson and Mace, 1975; Coughlin, 1976; Moser, 1984; Robens Institute, 1987). This chapter reviews available literature regarding perception to coastal pollution and the need to consider aesthetic quality of the coastal zone, in particular seascapes.

Extensive research has been carried out in relation to pollution perception of fresh waters, including lake and river water quality (Ditton and Goodale, 1973; Hertzgog, 1985; Burrows and House, 1989; House and Sangster, 1991; Smith *et al.,* 1995a), but little on coastal waters (Phillip, 1990, 1994a; Green and Birchmore, 1993; Smith *et al.,* 1995b). There is a lack of cohesion in this field of research, being developed by interdisciplinary schools of thought such as environmental psychology and environmental perception. Saarinen (1976) has commented on the distinct need for a defined methodology under an agreed heading to build a comprehensive body of theory.

4.b.2 Historical Development of Water Quality Standards

The World Health Organisation (1994a) identified 4 main and interdependent areas of potential environmental impact: physio-chemical, ecological, aesthetic and socio-economic. Historically, research and resulting legislation has been aimed at developing water indicators for guarding public health (Cabelli, 1979; Fleisher, 1990; Pike, 1994). Rees (1996) argued that visible signs of pollution rarely constitutes a specific health risk. However, as discussed and will be expanded upon later, there are various grounds for investigating environmental degradation and developing aesthetic indices which include moral, economic and tourism factors (David, 1971; Nicolson and Mace, 1975; House, 1986).

The EC Directive concerning the quality of bathing water (CEC, 1976) does target aesthetic indicators by setting Imperative standards, including colour, mineral oils, surface active substances and transparency, but only stipulating a Guideline set for floating materials. Certain derogations have been granted to the UK due to naturally turbid waters, eliminating the need for colour and transparency testing. In contrast, north American standards, always at the vanguard on water quality legislation, have more definitive and tighter controls on visual appearance of the marine offshore. Recreational water quality guidelines set to help establish relevant criteria in Canada for aesthetic value must be free from:

1. visible materials settling to form objectionable deposits
2. floating debris, oil, scum, and other matter
3. substances producing objectionable colour, odour, taste and turbidity
4. substances or combinations producing undesirable aquatic life
   (WHO, 1994a p.6)

All forms of aesthetic degradation need to be identified and causes pinpointed for appropriate action. At present there is a lack of data available to make accurate judgements on measuring visual pollution, which can take a variety of shapes impairing clarity, colour, surface quality such as slicks (Green and Birchmore, 1993). Poor aesthetic quality is not only attributable to anthropogenic causes resulting in floating debris and colour and clarity degradation due to sewage related debris, oil slicks, surfactants and general litter, but also due to natural phenomenon. For example presence of algal blooms can cause water discoloration in coastal waters and also health problems in humans in addition to fish kills (Rees, 1996), rotting seaweed has potential to create a pungent odour, a problem faced in Jersey, Channel Islands (Anon., 1994).

4.b.3 Research on Perception and Aesthetics

When considering aesthetics of the coastline it is important to investigate the source of contamination and to differentiate between marine and land based pollution. The main sources listed below contribute to both, either directly or indirectly:

- Sewerage systems
- Combined sewer outfalls
- Agricultural run off
- River/estuarine input
- Marine borne
- Deposited by recreationalists

A dynamic interaction occurs at the interface between land and sea, where a debris exchange takes place, particularly within the tidal area (Williams, *pers.comm.,* 1997). In addition other environmental forces such as the wind contribute to the process. The static nature of land allows applicability of cleansing techniques to clear debris. The volatile marine environment makes cleaning operations very difficult, if not impossible. The feasible solution is to control pollution from source.

From studies already carried out, strong similarities exist between perception of coastal aesthetics and the relationship between land and sea. Nicolson and Mace (1974) found the second and third most offensive forms of water pollution were murky dark water (26%) and floating debris (17%) which agree with David's (1971) findings reporting the same order of criteria but with murky dark water at (35%) and floating debris (20%). Nicolson and Mace (1974) also found that over 90% of respondents perceived water pollution purely on a visual basis, less that 10% mentioned non-visual indicators. A very strong link also appeared to exist between the presence of litter and perception of water quality. David (1971) reported perception of water quality was adversely affected by increasing quantity of litter, while Morgan *et al.* (1993) found on the Glamorgan Heritage Coast a positive correlation between perceived water cleanliness and absence of pollution backed by similar results from studies on the Turkish coastline (Morgan *et al.,* 1995). Both investigations were in accordance with results obtained by Everard (1995) and Dinius (1981). David (1971) also suggested turbidity has a strong influence on the way water is perceived. This was in consonance with Moser's (1984) work, that murky water or algae was indicative of bacterial or other harmful contaminants. The WHO (1994a) similarly stated that poor aesthetics of the water and surrounding environment can imply poor microbiological and chemical quality. A close tie can be seen between David's results (1971) and those of Dinius (1981) who found that the laymen's perception of discoloured water be indicative of marine pollution, and clarity to be significant in peoples perception of clean water (Herzgog, 1985; Burrows and House, 1989). The conclusion drawn by Ditton and Goodale (1974) was that people consider water dirty if not clear, when some waters are naturally turbid. Conversely water can be perceived as clean if clear. These findings are at variance with Smith *et al.* (1995a) in New Zealand who noted turbidity not to be an indicator of polluted waters, the significant factor being colour. Their results showed that turbid brown water was not regarded as suitable for bathing. However, in a later study (Smith *et al.,* 1995b), they substantiated the theory that colour was an important factor in water quality perception, but that overall site ranking suitability depended strongly on perception of visual clarity and not actual clarity. Duplicate findings were also produced by the Robens Institute (1987) supporting the notion that perceived cleanliness is reliant upon visual appearance.

Moser (1984) investigated public perception of water pollution in conjunction with levels of objective water quality. He found pollution to be generally judged less serious than from an actual biological point of view, many people being tolerant of water pollution. In an overall assessment of water quality, Moser (1984) noted colour, presence/absence of algae, presence/absence of floating debris, odour, movement and clarity to be criteria mostly used to describe the condition of the water. House and Sangster (1991) found comparable results showing strong association between perceived water quality and the presence/absence of individual water variables. They found smell, unusual colour and clear water to be important in the public's evaluation of water quality supporting earlier work by Burrows and House (1989). More recent work done by House (1995) and House and Herring (1995) investigated the perception of sewage related waste, on public enjoyment. Debris from sewage proved to be the most offensive form of aesthetic pollution. Williams and Morgan (1995) also found sewage to have the most detrimental effect on beach enjoyment by the public. Their results indicated that the most desired beach qualities were absence of sewage debris, oil and litter as well as clean bathing water. From a list of overall coastal characteristics, 60% of beach users placed highest priority on clean sand and water.

A number of studies found a strong relationship between water appearance and bathing activity. This correlated well in studies by Smith *et al.,* (1995b) and Morgan *et al,* (1993) who showed a close association between perceived water cleanliness, absence of pollution and quality of the beach for swimming. Phillip (1994a) and the WHO (1994a) elsewhere reported that poor aesthetic appearance of bathing water and bathing beaches, in particular with relevance to specific items, have shown a positive correlation with higher rates of self-reported gastrointestinal illness after swimming in sewage polluted water. These items are:

1. discarded food/wrapping
2. bottles/cans
3. broken bottles
4. paper litter
5. dead fish
6. dead birds
7. chemicals
8. oil slicks

9. human/animal excrement
10. discarded condoms
11. discarded sanitary towels.
(Phillip, 1994a p.5).

4.b.4 Socio-Economic Factors

A broad brush picture has been painted highlighting public perception to aesthetic pollution. Any attempt at dealing with re-occurring common problems from the various studies must acknowledge the widely varying types of water-scape, beach type and contextualise the coastal environment in terms of user. The beach consumer must be broken down into a number of descriptor variables, inclusive of age, gender, socio-economic type, proximity to beach and activity type. Ditton and Goodale (1973) noted locals were mostly likely to perceive pollution in contrast to visitors. However, Cutter *et al.* (1979) found a trade-off appeared to exist between convenience and quality. The inference was that local people would sacrifice choice of beach destination and quality for accessibility. Ditton and Goodale (1973) also observed a difference in perception of user type. In their study of Green Bay, Lake Michigan, boaters were most tolerant of pollution, swimmers least tolerant and fishermen somewhere in between the two. Results from a study by Moser (1984) contradict these findings, concluding that estimates of quality are not affected by water-related activities. It must be borne in mind that it is not just the water recreationalist who is affected by pollution or more accurately visual impairment of water body. The amenity of a water body must also be considered in terms of non water contact activities such as walking and picnicking (Burrows and House, 1989). Fisher and Raucher (1984) went further by measuring the benefits of non-use in relation to water quality, concluding the importance of intrinsic benefits in terms of ecology, need for others, aesthetics and future use. Williams *et al.,* (1995) found females to be less tolerant of pollution than males and they also found that beach users from a higher socio-economic class tended to perceive poorer water quality than lower groups, confirmed by studies by Young *et al.* (1996) and Morgan *et al.,* (1993). Age was another important factor to be considered. Although no specific results were found regarding tolerance to pollution, differences in preference to beach type existed, older people choosing quieter less populated beaches (Anantharaman, 1980).

4.b.5 National and Supranational Recognition: Initiatives Evaluating Aesthetic Problems of Recreational Waters

Attempts are being made to address the problems of aesthetic pollution and develop methodologies through the identification of appropriate indicators. Phillip (1990, 1994a) has carried out work in this field in connection with the WHO Regional Office for Europe, looking for different health indicator methods to help appraise quality of bathing water and bathing beaches. Listed are the requirements of aesthetic indicators, being able to:

a. classify different levels of beach and water quality before and after any cleansing

b. be useful when compared with conventional bacteriological and chemical indicators of recreational water quality and the likelihood of illness amongst different groups of recreational water users.
   (Phillip, 1994a p.9).

Initiatives to protect the environment from aesthetic degradation are being put into motion on both a national and supranational scale. The European Charter on Environment and Health acknowledged in Principles for Public Policy, point 1:

> *'Good health and well-being require a clean and harmonious environment in which physical, psychological, social and aesthetic factors are all given their due importance'.*
> (WHO, 1989a p.4)

The Charter also identifies the necessity for inter governmental collaboration on common environmental and health issues. Co-operation across intra-governmental agencies is also essential. Local Agenda 21 (Harman *et al.,* 1996) points out the importance in terms of sustainable development the appreciation of managing the environment hand in hand with social and economic issues, setting the ground for local authority action. In support the British Government White Paper, 'The Health of the Nation' (DoH, 1992), noted the necessity to accurately recognise the connection between health and the quality of the environment.

4.b.6 Tourism

Identification of pollution problems on the coast and impact of aesthetic degradation on tourism needs careful consideration. Rees (1996) outlined the effect on amenity value with reduced transparency, discoloration, scum-foaming and off smells, which can ultimately lead to a loss income to localised areas. Phillip (1990) has recognised that aesthetic concern for recreational water quality can have profound economic effects. Work done by the WHO (Phillip, 1994a) listed the detrimental economic effects on tourism from mis-management of the marine environment:

- number of tourist days lost;
- damage to the local tourist infrastructure (hotels, restaurants, resorts etc.)
- damage to tourist-dependent activities (food industry, general commerce etc..)
- damage to fisheries activities (stoppage of fisheries, depreciation of fish price)
- damage to fisheries dependent activities (fishing equipment production and sales)
- damage to image of the Adriatic Coast (or any coastal region) as a recreational resort at both national and international levels.

(Phillip, 1994a p.8)

Nicolson and Mace (1975) identified the need to take a holistic view of the situation and noted that water quality may be an important factor in the influence of user destinations and tourism. They also recognised that protection of the water body be based on economics and cost benefit of a particular area. Burrows and House (1989) also reported on the importance to investigate the water body in terms of usage, for cost benefit analysis. Grant and Jickels (1995) make it clear that although beach cleansing operations are essential in tourism terms, the potential high cost to the local community be fully appreciated. David (1971) suggested pollution be put into perspective by measuring resultant changes over time, or identifying whether a point source reaction to an individual event has occurred, with temporary consequences.

4.b.7 Landscape and Planning

The outdoors, especially the coast are a vital resource offering a perfect environment for a diverse range of activities from sports and recreation to providing a unique place for relaxation and peacefulness. Williams and Lavelle (1990) commented that the most common aspect of public enjoyment of the outdoor is perception of the landscape. Smith *et al.,* (1995b) also suggested in a coastal context the surround of water creates a feeling of pleasantness, acknowledged by Herzgog (1985) who highlighted the importance for decision makers in understanding the cognitive process involved in evaluating waterscapes. It would therefore be prudent to be aware of the user's perception of landscape and seascape in the planning process. Burton (1971) substantiated this theory by arguing that the social role of attitude and perception studies be incorporated in the planning process. Local Authorities must also be aware of the perception of the user in any decision making process, and ensure dissemination of information outlined by the WHO (1994a). Ditton and Goodale (1974) make clear that the contrast between science and perception has received little attention. The public are unable to understand terms such as turbidity and coliforms. The implication is that understandable data should be displayed for public access.

4.b.8 Management

In the past an intellectualised approach has been employed setting standards for beach management. Examples of this can be seen with the introduction of the European Blue Flag, and in the UK the design of the Tidy Britain Group Seaside Awards. The process needs to be reversed and management focused around principles evolved from a bottom up approach. Recent work is developing based on views of the user, such as Morgan's (1996) beach rating scheme. However, Williams *et al.* (1993) acknowledged the inadequate supply of research in this field. Understanding the cognisance of the end user is imperative in evaluating different land and waterscapes to assess the impact of aesthetic degradation (Coughlin, 1976; Herzgog, 1985) and appreciate the intrinsic benefits of the coast in ecological and aesthetic terms (Fisher and Raucher, 1984).

Nicolson and Mace (1975) noted that physical parameters might not bear a relationship to demand, but perception and tolerance understanding are critical. Phillip (1994a) stated that all factors including absorption capacity, economic, ecological and human aspects should be accounted for in an overall management strategy. Nicolson and Mace (1975) were in agreement with these views but stated that whilst political, financial and environmental carrying capacity all need to be managed, the importance of perception must now be recognised. Identification between natural and anthropogenic inputs are an essential part of good management practice. Where naturally discoloured or turbid waters, which detrimentally effect the perception of the beach consumer exist with no adverse risk to health, localised education programmes should be implemented (Smith *et al.,* 1995a). Dinius (1981) also acknowledged the necessity to understand the relationship of perception to colour and clarity for effective coastal zone management.

The regulatory authorities are slowly becoming to realise the importance of the more subjective areas involved with management of the coast. The WHO (1990a) outlined the need for development of appropriate aesthetic indicators to monitor and aid control of visual quality of the coastline. Nicolson and Mace (1975 p.1207) noted the necessity to consider all aspects of water quality, quoting 'recreational, social and economic variables must be related to environmental quality so that effective management procedures can be implemented to control and improve water quality'. Finally any developed plans should ensure that the beach user is given accurate and understandable data on which to make their own decision and judgement.

Chapter 5 Methods and Data Analysis

5.1 Methods for Measuring Coastal Water Quality at Whitmore Bay

5.1.1 Water Sampling Technique

5.1.1.1 Sampling Sites

The NRA Bathing Water Sampling Procedure (1995) was followed for water quality analysis at Whitmore Bay. Microbiological samples were taken over two bathing seasons during 1995 and 1996. Two sample sites were chosen for water analysis in order to obtain data across the Bay width, in contrast to the Environment Agency who only sample at one point (Environment Agency, 1997). The first site S1, central bay, is the same position used by the Environment Agency, for comparison purposes (OS GR: ST 115 662), Figure 2.1. Site 2 (S2) is 300 metres directly west of site 1, chosen in preference to the eastern side of the beach due to higher density of bathers. The purpose of sampling in 1996 was to verify the water quality analysis conducted in 1995, and not for inclusion in the statistical modelling on health risk. For this reason, replicate sampling was carried out in 1996, taking three samples from the central location at each sampling time.

5.1.1.2 Frequency

The pilot study highlighted a window period between 11.00am and 3.00pm as being the sampling optimum time, i.e. the time of highest bathing load at the beach. Therefore, the sampling programme was designed around this time to ensure the microbiological analysis was representative of the primary swimming period. During the 1995 survey, samples were taken at 11.00am, 1.00pm and 3.00pm from both sampling stations S1 and

S2. The same time frame was used for the 1996 survey, but as mentioned only S1 was sampled.

The survey for 1995 was conducted over a six day period and due to restrictions with limited laboratory capacity, frequency of sampling was employed in preference to confirming presumptive results. The survey for 1996 was conducted over a three day period. Although a comprehensive 24hr picture of microbiological densities on the survey days was not represented in the results, fluctuations over the complete spectrum of the tidal cycle were recorded.

5.1.1.3 Procedure

All field observations were recorded, including date and time, along with prevailing environmental conditions and tide state. Samples were taken as close to the sampling site as possible for continuity and comparison of results. The 1.5 litre sample bottles had a screw top and were sterilised before sampling. Precautions were taken to avoid exogenic contamination. A 2m sampling rod was used to distance the bottle from the sampler, the clamp sterilised with medical wipes and the sampler wore disposable gloves (WHO/UNEP, 1994). At each sampling station the sampler fixed the water bottle to the clamp, removed the screw top, avoiding contaminating the mouth of the bottle with the gloves and gently waded to approximately knee depth of water trying to avoid disturbing the seabed sediment. The sampler then extended the sampling rod seawards and submerging the sample bottle inverted to a depth of 30cm (Robens Institute, 1993; CEC, 1997). The container was then turned through 180^0 with the mouth facing the current, away from the sampler, again avoiding sampler contamination, and filled leaving a gap of 20mm at the top to allow for mixing before analysis (HMSO, 1994). The bottles were made of borosilicated glass (WHO/UNEP, 1994).

5.1.1.4 Storage

Immediately after sampling, samples were transferred to a thermoisolated box, away from light and transported straight to the laboratory. All sample bottles were pre-labelled for reference. Analysis of *E.coli* and faecal streptococci were performed within 4-6hrs of sampling (HMSO, 1994). Samples tested for bacteriophages were refrigerated overnight and then transported in a thermoisolated box, protected from light, to acer Laboratories, Bridgend, South Wales.

5.1.2 Microbiological Analysis

Waterborne pathogens occur in natural waters due to discharge of sewage or wastewater. Sewage contains a high density of pathogenic micro-organisms, which potentially cause a health hazard when released into recreational waters. The diverse nature of waterborne pathogens makes detection difficult. Therefore, indicator organisms have been developed to monitor water quality and indicate the presence of sewage and thus the likelihood of pathogenic micro-organisms.

The two prime indicator organisms stipulated in the EC Bathing Water Directive (CEC, 1997) are *E.coli* and faecal streptococci, which have formed the basis for this investigation. The Directive has also created provision for future inclusion of bacteriophages (phages) to indicate presence of sewage, dependent on future research to isolate an appropriate phage (Nelson *et al.,* 1997). Total coliforms were originally included in the first EC Bathing Water Directive (CEC, 1976), but have since been removed due to their natural occurrence everywhere in nature (EC, 1995). The WHO (1991) report on health risks from bathing in marine waters also prescribe the use of *E.coli* and faecal streptococci as suitable indicators for epidemiological-microbiological investigations. In addition the WHO/UNEP (1989b) set out a protocol for assessing water quality of recreational waters.

5.1.2.1 Microbiological Technique

The purpose of microbiological methods are to detect and/or enumerate particular micro-organisms, i.e. the target organisms (EC, 1995). Other micro-organisms may be present, but should go undetected and should not interfere with the analytical process (EC, 1995). The two main microbiological techniques for indicator organism enumeration which are most widely used (Fleisher, 1990b), Membrane Filtration (MF) and Most Probable Number (MPN), are both acceptable under the EC Bathing Water Directive (CEC, 1997). The MF procedure provides actual estimates of indicator organism densities, whereas the MPN procedure provides a statistical derived estimate of indicator organism densities. Moreover, the MF is a more precise method (Fleisher, 1990a) and major advantages of MF are the speed with which results can be obtained as direct counts and the relatively low cost in terms of labour, media and glassware compared to other techniques (HMSO, 1994). The main disadvantages of MF are that it is not suited to waters which are highly turbid, with sediment blocking the membrane and inhibiting growth or for samples which produce low counts (HMSO, 1994). Even though the seawater at Whitmore Bay is turbid, it did not interfere with the microbiological analysis, especially at low dilutions, which usually produced sufficient counts.

The standard MF technique undertaken for bacterial analysis in this investigation is detailed in the HMSO publication Report 71 (HMSO, 1994). In summary the MF technique involves filtering a known volume of water through a membrane filter, which retains the micro-organisms. The membrane filter is then placed on a solid medium, which is generally selective. During incubation, at a set temperature and time period, (depending upon the specific germs) the micro-organisms develop into visible colonies. The number of Colony Forming Units (CFU) can be identified, and a value given to the sample. Depending on the dilution, the CFU is multiplied up to give a value in terms of 100ml. For the analysis of both *E.coli* and faecal streptococci 1ml, 10ml and 100ml dilutions were made. Geometric means were calculated to describe the data (Fleisher, 1990b), which generally involves the transformation of the data to log_{10} values. This transformation reduces the likelihood of abnormally high or abnormally low counts in a small number of samples having undue influence on the overall mean of a large series of observations (Robens Institute, 1993). The WRc (1996a) also used geometric means to re-analyse the data of the Beach Surveys and Cohort Studies carried out between 1989 and 1992 (Pike, 1994).

5.1.2.2 Thermotolerant (Faecal) coliforms

Faecal coliforms have all the characteristics of coliforms, but are able to ferment lactose with the production of gas in 24 hrs at 44^0C (Dufour, 1977). Faecal coliforms are used to denote a coliform of faecal origin and capable of growth at 44^0C, i.e. thermotolerant. Faecal coliforms belong to the family *Enterobacteriaceae*, and mainly consist of *Escherichia* (*E.coli*) and *Klebsiella* (cited Furlong, 1996). *E.coli* is the only coliform which is known to definitely inhabit the gastrointestinal tract (Dufour, 1977) and thus the EC Bathing Water Directive (CEC, 1997) has refined its determinand for faecal coliforms to *E.coli*. At the time of this study the criteria stated in the original EC Bathing Water Directive (CEC, 1976) was to investigate the presence of faecal coliforms, which were tested for in both the 1995 and 1996 surveys.

Membrane filtration was onto lauryl sulphate broth and resuscitated at 30^0C for 4 hours; after resuscitation incubated at 44^0C for 14 hours (Robens Institute, 1993; HMSO,1994). The CFU were yellow in colour (HMSO, 1994).

5.1.2.3 Faecal streptococcus

Faecal streptococci are a heterogeneous group of organisms and are always present in the faeces of warm blooded animals (Knudson and Hartman, 1992). The EC Bathing Water Directive (CEC, 1997) defines faecal streptococcus as corresponding to a heterogeneous group of the genera *Enterococcus* and *Streptococcus*. The original EC Bathing Water Directive (CEC, 1976) only included a Guideline standard for faecal streptococci, but more recent research has shown its usefulness as an indicator of sewage, correlating with rates of gastrointestinal (Kay *et al.,* 1994) and an Imperative standard has been introduced into the amended EC Bathing Water Directive (CEC, 1997).

Membrane filtration was onto Slanetz and Bartley agar and resuscitated at 37^0C; after resuscitation incubated at 44^0C for 44 hours (Robens Institute, 1993; HMSO,1994). The CFU were pink, red and maroon in colour (HMSO, 1994).

5.1.2.4 Bacteriophages

Bacteriophages are micro-organisms composed of a virus attached to a bacteria (Scarpino, 1978; Hugo, 1964). Many phages have a similar response to the environment as human viruses and are relatively easy to detect using simple, rapid and inexpensive methods, making them a viable substitute for enteroviruses (Fewtrell and Jones, 1992). No standardised protocol exists to analyse phages under natural conditions in sewage and receiving waters and little is know of their densities therein or in human faeces (EC, 1995). Havelaar (1993) has suggested F-specific RNA bacteriophages as an appropriate model for enteroviruses in bathing waters. In 1995, inclusion of bacteriophages into the reformed EC Bathing Water Directive was imminent and Rees (*pers.comm.*, 1995) proposed F-specific RNA phages were tested for in the microbiological analysis. The laboratory at the University of Glamorgan did not have the level of standard required to test for bacteriophages, so acer Laboratories (Welsh Water) carried out the analysis. They used their own procedure, which was not made available. The amended EC Bathing Water Directive (CEC, 1997) has made provision for future inclusion of bacteriophages into the Directive, based on further epidemiological research.

5.1.3 Secchi Disc Measurements

Turbidity is the result of suspended matter in the water column, both organic and inorganic. These particulates consist mainly of suspended microscopic plants and animals, suspended mineral particles, stains that impart a colour, detergent foams, dense mats of floating and suspended debris, clay, silt or a combination of these factors (Internet 1996b; Phillip, 1996). Turbidity in the Bristol Channel consists mostly of silt originating from the Severn Estuary (Severn Estuary Strategy, 1997). Suspended matter both absorbs and scatters light (Pilgrim, 1984), attenuating light penetration into the water column, which has detrimental effects on survival of aquatic organisms (Internet, 1996a) and degrades the aesthetic quality of recreational waters (Phillip, 1996). Turbidity is measured as the intensity of light scattered at 90^0 to the path of the incident light, and given in nephelometric units, or as parts per million (ppm) (Pilgrim, 1984).

A Secchi disc measures transparency which is a function of turbidity and Secchi disc depth (SD) is inversely proportional to increasing suspended sediment in the water column (Pilgrim, 1984; Internet, 1996a). As turbidity represents relative clarity of water (Internet, 1996b) and therefore inter-related with transparency it has been used in this study to describe the clarity of recreational waters. A Secchi disc is perhaps the oldest tool used for measurement of water clarity and it is a cheap and simple instrument that gives an immediate indication of water turbidity (Pilgrim, 1984; Carlson, 1995). The EC Bathing Water Directive stipulates a both a Mandatory standard SD of 1m and Guideline standard SD of 2m for European bathing waters (CEC, 1997). However, the Welsh region has a derogation for transparency due to naturally turbid waters and the Environment Agency do not carry out physical turbidity tests due to dangers inherent with poor visibility of the sea bed, but instead use a visual check (Roberts *pers.comm.,* 1996).

The size of a Secchi disc for measuring marine waters is approximately 400mm and for fresh waters approximately 200mm, and is divided in to two white quadrants and two black quadrants (Francis *et al.,* 1994; Carlson, 1995; Internet, 1996b). The internet gives a full description on how to construct a Secchi disc (Internet, 1996a, 1996b). The disc works on contrast between the white quadrants and the black background of the bed of the recreational waters and disappears when the human eye no longer sees it. But when the bottom is not totally black, the white quadrants disappear from view sooner than would be expected and under these circumstances the black quadrant provides a black background standardising the contrast (Carlson, 1995).

Most of the literature addresses turbidity of fresh waters (Anon., 1987; Francis *et al.,* 1994; Carlson, 1995; Phillip, 1996), but the operation of a Secchi disc is the same. The Secchi disc is lowered into the water on a graduated line and the point at which it disappears reordered. The Secchi disc is then raised and the point at which it reappears recorded. The SD is the average value of the two readings (Pilgrim, 1984; Carlson, 1995; Internet 1996a, 1996b; Phillip, 1996). Francis *et al.,* (1994) found all criteria from his results met parametric statistics in line with these findings. Secchi disc readings were taken at three equidistant positions at all three beaches, at different times to record both spatial and temporal variations.

5.2 Statistical Techniques for Health Risk Assessment

Aetiological studies require rigorous statistical analysis to verify findings. A series of statistical tests were applied to the epidemiological data in order to establish whether a relationship existed between exposure to sewage contaminated sea water at Barry Island and health risk, and also to investigate whether there was evidence of a dose response relationship between illness and faecal bacteria. In a first stage observation to determine if an association between illness contraction and immersion in sea water existed, chi-square (χ^2) analysis was performed (Siegel, 1956; Jandel Scientific, 1995). A second phase analysis using odds ratios (Hosmer and Lemeshow, 1989) was performed to establish the relative risk of illness between the exposure group (cases) and the non-exposure group (controls). Stratification of the data set allowed comparison of risk values across independent variables such as age and gender. However, the odds ratio whilst providing risk statistics for the stratified set of data does not provide control for confounding and interaction effects, discussed in more detail below. To account for this the Mantel Haenszel Method (Schlesselman, 1982) was investigated which computes a weighted summary estimate of risk. The final method employed was multiple logistic regression, a powerful statistical technique which generates odds ratios in the presence of confounding or interaction effects (Breslow and Day, 1980; Collett, 1991).

5.2.1 Test of Association Between Exposure and Illness (χ^2)

The Chi-square (χ^2) test is suitable for non parametric data and can be applied to contingency tables. The test was used to determine if an exposure/effect association existed between the control group of non swimmers compared to the swimming group. The statistic χ^2 is defined as (Siegel, 1956):

$$\chi^2 = \sum \left\{ \frac{(\text{Observed - Expected})^2}{\text{Expected}} \right\}$$

The null hypothesis H_0 states that no association exists between variables, that is they are independent. Conversely the alternative hypothesis H_A states that the variables are associated. The size of the contingency table determines the number of degrees of freedom, and tests were carried out to a level of significance $P = 0.05$.

5.2.2 Odds and the Odds Ratio (ψ)

There are different methods for measuring degrees of association between exposure and health risk. Measures of association try to establish the extent to which levels of exposure and disease are related. Further detailed explanations of risk can be found in Collett (1991) and Schlesselman (1982).

The simplest measurement of **risk** describes the probability of new cases of illness occurring as a proportion of the population at risk over a given period of time. ***Relative risk*** of disease is a ratio which measures the likelihood that a person exposed to a particular factor is to a greater or lesser extent at risk of contracting a disease compared to someone who has been unexposed. If P_e represents the risk of disease occurring in the exposed group and P_u in the unexposed group then the relative risk ϱ is:

$$\varrho = \frac{P_\varepsilon}{P_u}$$

In the case of many epidemiological studies including this one the outcome variable is dichotomous. The **odds** and **odds ratio** (ψ) are useful when considering a binary response variable. Odds of success is defined as the ratio of the probability of success (P) over the probability of failure (1- P):

$$Odds\ of\ success = \frac{P}{(1-P)}$$

When two sets of binary response data exists, for example between an exposed group and an unexposed group the relative odds is equal to the ratio of the odds of success for the exposed over the odds of success for the unexposed (Collett, 1991). This is known as the odds ratio denoted by ψ, such that:

$$\psi = \frac{P_e/(1-P_e)}{P_u/(1-P_u)}$$

If the odds are identical then ψ is unity. If the odds of success are higher in the exposed group ψ will be greater than one and conversely if the odds of success are greater in the unexposed group then ψ will be less than one.

In the context of this study odds of contracting an illness needs to be expressed as a ratio between the exposed group, which are those respondents in the survey that entered the water compared to the control group of unexposed participants, who *did not* enter the water. The odds ratio for the exposed and unexposed groups can be calculated from a 2x2 contingency table, Table 5.2.1 (Hosmer and Lemeshow, 1989):

	Ill	Not Ill
Exposed	*a*	*b*
Unexposed	*c*	*d*

Table 5.2.1 Exposed vs. Unexposed

$$\varrho = \frac{P_e}{P_u} = \frac{a(c+d)}{c(a+b)}$$

$\{ P_e = a/(a+b); P_u = c/(c+d) \}$

$$\psi = \frac{P_e/(1-P_e)}{P_u/(1-P_u)} = \frac{ad}{bc}$$

$$\psi = \frac{ad}{bc}$$

The approximate confidence limits indicate the reliability of the odds ratio. The odds ratio is significantly different from unity if the confidence interval does not include unity. Upper and lower 95% confidence limits are given by Woolf (1959), cited Schlesselman (1982).

$$\psi_{\text{lower limit}} = \psi \exp [-1.96 \sqrt{\text{var} (\ln \psi)}]$$

$$\psi_{\text{upper limit}} = \psi \exp [+1.96 \sqrt{\text{var} (\ln \psi)}]$$

$$\text{var} (\ln \psi) \sim (1/a + 1/b + 1/c + 1/d)$$

The odds ratio obtained from the 2x2 contingency table give a crude estimate of the illness rates from exposure to seawater. However, this analysis does not take account of confounding or interaction effects between potential risk factors (e.g. age and gender) and illness. By stratifying the data by age for example, odds ratios for different age groups can be calculated and if large differences are observed in these odds ratios, then the presence of confounding or interaction may be suspected. The Mantel- Haenszel Method is designed to statistically adjust for confounding effects. However, the technique of logistic regression enables one to control for the presence of confounding and interaction effects in the calculation of the odds ratio.

5.2.2.1 Confounding and Interaction

In an epidemiological study of this nature it is necessary to consider and account for the potential effects of interaction and confounding. In the first set of analysis the principal independent risk variable, or exposure variable, under investigation was exposure to sea water, discriminating between swimmers and non-swimmers. The second stage of analysis concentrated only on those that entered the water. The main driving variables in this set were *E.coli* and faecal streptococci, attempting to establish whether a dose response relationship existed between concentration of faecal bacterial indicator organisms and health risk. A variable which is associated with the driving variable(s), which either elevates or reduces the risk of infection but is not a consequence of exposure is said to have a confounding effect. Hosmer and Lemeshow (1989 p.63) give an exact definition of confounding, which will become more apparent in view of the subsequent logistical analysis:

> *'the term confounder is used by epidemiologists to describe a covariate that is associated with both the outcome variable of interest and a primary independent variable or risk factor. When both associations are present then the relationship between the risk factor and the outcome variable is said to be confounded. The procedure for adjusting for confounding is appropriate when there is no interaction present. Confounding is present when the addition of a variable to the model produces significant changes in the existing regression coefficients'.*

In simpler terms, the statistical analysis chosen is required to give an accurate estimate of true risk attributed strictly to the exposure variable controlling for confounding factors.

When calculating an odds ratio across stratified subgroups ψ must represent the constant component of association. Schlesselman (1982) stated that the apparent odds ratio or relative risk may give an indication of whether an exposure effects the risk of disease, but the magnitude may be over or underestimated. Failure to acknowledge the importance of statistically adjusting for confounding variables will ultimately lead to erroneous results.

Interaction occurs if the degree of association between the risk factor and outcome variable is different within each level of the covariate. Collett (1991) defined interaction as occuring when a confounding variable modifies the effect of the exposure factor on the disease. The Mantel Haenszel Method does not account for the effects of interaction, however, logistic regression can be used to calculate the revised odds ratio in the presence of interaction.

5.2.3 Mantel-Haenszel Method (ψ_{mh})

The Mantel-Haenszel Method is an efficient technique for estimating a summary odds ratio from a series of contingency tables. The method computes a weighted summary odds ratio (ψ_{mh}) controlling for variables shown to influence the effect of exposure, i.e. confounding factors.

If the exposed and unexposed categories are divided into k subgroups, the observations in the subgroups regarded as the i^{th} terms and n representing the sum total of values a,b,c and d in the 2x2 contingency tables the Mantel-Haenszel summary estimate of the odds ratio is calculated as (Schlesselman, 1982):

$$\psi_{mh} = \sum^{k}_{i=1}(a_i\, d_i \,/\, n_i) / \sum^{k}_{i=1}(b_i\, c_i \,/\, n_i)$$

The null hypothesis H_o states that no exposure disease relationship exists, i.e. H_0: $\psi_{mh} = 1$

Approximate upper and lower confidence limits for ψ are calculated by taking antilogs of the upper and lower limits for ln ψ:

$$\psi_L = \psi_{mh} \exp [-z_\alpha \sqrt{\mathrm{var}(\ln \psi_{mh})}]$$

$$\psi_U = \psi_{mh} \exp [+z_\alpha \sqrt{\mathrm{var}(\ln \psi_{mh})}]$$

Note the variance is calculated thus:

$$\mathrm{var}(\ln \psi_{mh}) \cong \sum \varpi^2_i \upsilon_i / (\sum \varpi_i)^2$$

$$\varpi = b_i c_i / n_i$$

$$\upsilon = (a_i + c_i) / a_i c_i + (b_i + d_i) / b_i d_i$$

Two main problems were encountered using the Mantel-Haenszel Method (ψ_{mh}). Firstly the method breaks down if zero entries occur for either values b_i or c_i, and secondly the method fails to take account of interaction. To overcome this fewer strata must be used or approximate methods utilised (Schlesselman, 1982)

5.2.4 Multiple Linear Logistic Regression

Multiple logistic regression (MLR) is a powerful statistical modelling tool ideally suited for analysing epidemiological data (WHO/UNEP, 1991) which is characterised by a qualitative binary dependent variable, for example the presence or absence of disease (Breslow and Day, 1980; Cox, 1989; Hosmer and Lemeshow, 1989; Internet, 1996). The model is designed to describe the relationship between the mean response, in this study the contraction of illness, and one or more explanatory variables. For a detailed explanation on MLR refer to Hosmer and Lemeshow (1989) and Collett (1991). The following notes attempt to summarise the MLR technique. They were formulated in conjunction with Richards (*pers.comm.,* 1996) and based on the work by Collett (1991).

5.2.4.1 The Linear Logistic Model

The logistic model can be linearised by taking the logit transformation. Suppose we have n binary observations y_i, $i = 1,2,\ldots n$ and P_i is the success probability corresponding to the i^{th} observation. The linear logistic model for the dependence of P_i on the values of the k explanatory variables $x_{1i}, x_{2i}, \ldots x_{ki}$ associated with that observation, is

$$\text{logit}\ (P_i) = \log\ [P_i/(1-P_i)] = \beta_0 + \beta_1 x_{1i} + \beta_2 x_{2i} + \ldots \beta_k x_{ki}$$

which on re-arrangement gives

$$P_i = \frac{\exp\ (\beta_0 + \beta_1 x_{1i} + \ldots \beta_k x_{ki})}{1 + \exp\ (\beta_0 + \beta_1 x_{1i} + \ldots \beta_k x_{ki})}$$

where $\beta_0 + \beta_1 + \beta_2 + \ldots \beta_k$ are unknown parameters and $x_{1i}, x_{2i} \ldots x_{ki}$ are known (Richards *pers.comm.,* 1996). Thus, it can be seen that the logit (P_i) is log (odds ratio) and is linearly related to the explanatory variables. Both theoretical and empirical considerations suggest that when the response variable is binary the shape of the response function will frequently be curvilinear. The relationship between P_i and the explanatory variables is said to be sigmoidal.

The linear logistic model can be used to predict the log (odds ratio) of contracting a disease for a given set of values of the explanatory variables. However, the emphasis of this investigation is to accurately estimate the odds ratio of contracting an illness from exposure to seawater in the presence of confounding and/or interaction. These estimates are obtained from the parameters β_j in the model.

5.2.4.2 Fitting the Linear Logistic Model to Binary Data

When dealing with binary data and utilising a linear logistic model the $(k+1)$ unknown parameters β_0, β_1, β_2,...β_k have first to be estimated. The method of Maximum Likelihood may be employed to estimate the β_j parameters of the logistic response function by deriving a set of $(k+1)$ non-linear equations (Collett, 1991). Standard numerical search procedures are used to find the maximum likelihood estimates which maximise the log-likelihood function, which are widely available in a number of statistical computer packages. Statistica for Windows (Statsoft, 1993), Jandel Scientific (1995) and SPSS(1995) were all used in the production of results for this thesis.

5.2.4.3 Representation of Variables

In many epidemiological studies including this one, multi-variable data is obtained and as a consequence linear logistic models fitted to data from such studies will generally include a number of different terms. The resultant data may include more than one exposure variable, confounding terms between the probability of disease and exposure factors and interaction effects between exposure and confounding factors. Health risk data collected for this research was based on studies by Lightfoot (1989), Alexander and Heaven (1990), Balarajan *et al.,* (1992 and 1993) and Jones *et al.,* (1993). The variables included age, sex, visitor status, socio-economic status, exposure three days prior to survey, head immersion, activity and consumption of certain risk foods, such as a burger. The exposure variables were Enter (whether or not the subject entered the water) and the bacterial indicator organisms *E.coli* and faecal streptococci.

It was necessary to code the data for use in the logistic model. Continuous data only occupies one variable with the corresponding value of the factor. For discrete qualitative data such as dichotomous and polychotomous, indicator or dummy variables were used (Hosmer and Lemeshow, 1989; Internet, 1996). The majority of predictor variables in this study had only two possible outcomes, taking on the value of a 1 or 0, for example in the instance of sex a male was coded 1 and females were coded 0. In the case of a categorical variable (polychotomous) having more than two possibilities, for example socio-economic status, for *n* categories *n*-1 dummy variables are required using one group as a reference. It is important to make clear that all dummy variables are needed to represent a particular level. Level one is used as a baseline to compare the remaining levels by setting all dummy variables to zero. Finally, interaction effects between two factors are represented by the product of the variables representing the factors (Richards *pers.comm.,* 1996). If X1 represents the factor sex and X2 represents the factor head immersion, then the variable X3 =X1 x X2 represents the interaction between sex and head. Further if X1 represents the factor sex and X2, X3, X4 and X5 represents the factor socio-economic status the variables X6 = X1 x X2; X7 = X1x X3; X8 = X1x X4 and X9 = X1 x X5 represents the interaction between sex and socio-economic status. The coding procedure used in this study was as follows:

Dichotomous Variables

i. Enter	0 = Not Enter	1 = Enter
ii. Sex	0 = Female	1 = Male
iii. Visitor	0 = Local	1 = Travel > 10miles
iv. Exposure 3 days previous	0 = No Exposure	1 = Exposure
v. Head immersion	0 = No	1 = Yes

vi. Activity	0 = Wade	1 = Swim
vii. Foods	0 = No Burger	1 = Burger

Polychotomous Variables

viii. Age	Level 1: age 40+ (reference group)	X1= 0, X2 = 0
	Level 2: age 0-19	X1= 1, X2 = 0
	Level 3: age 20-39	X1= 0, X2 = 1
ix. Socio-economic status	Level 1: employed (reference group)	X1= 0, X2 = 0
	Level 2: housewife and retired	X1= 1, X2 = 0
	Level 3: students and unemployed	X1= 0, X2 = 1

N.B. The socio-economic status group had to be collapsed due to low numbers in the contingency tables, which if not altered would have produced erroneous results.

Using the modelling process to investigate the effect of faecal indicator concentration on illness rates two sets of analysis were performed. The first analysis adopted a single variable to code *E.coli* and a single variable to code faecal streptococci. These variables took on two values to define a significant biological cut off point using the Mandatory standard for *E.coli* (2000 per 100ml) and Guideline standard for faecal streptococci (400 per 100ml), defined by the original EC Bathing Water Directive (CEC, 1976). At the time there was no Mandatory value set for faecal streptococci. The second analysis used dummy variables to accommodate all the daily geometric mean values calculated for both *E.coli* and faecal streptococci over the six day sampling period. The dummy variables EX1, EX2 ...EX5 were used for *E.coli* and SX1, SX2 ...SX5 for faecal streptococci, Table 5.2.2.

E.Coli/100ml	EX1	EX2	EX3	EX4	EX5
1052	0	0	0	0	0
1799	1	0	0	0	0
2257	0	1	0	0	0
3709	0	0	1	0	0
5507	0	0	0	1	0
17975	0	0	0	0	1
F.Strep/100ml	SX1	SX2	SX3	SX4	SX5
260	0	0	0	0	0
301	1	0	0	0	0
339	0	1	0	0	0
408	0	0	1	0	0
459	0	0	0	1	0
1491	0	0	0	0	1

Table 5.2.2 Coded Values for E.coli and Faecal Streptococci

5.2.4.4 Model Fitting

The objective of the analysis was to find the best fitting linear logistic model to accurately describe the relationship between the disease outcome and the exposure factors in the presence of potential confounding and interaction effects. Alternative linear logistic models can be compared in terms of a statistic called the deviance. When one model contains factors that are additional to those in another, the difference in the deviances of the two models measures the extent to which the additional factors improve the fit of the model to the observed response variable. The deviance D for binary data y_i, $i = 1,2,\ldots n$ was given by (Collett, 1991)

$$D = -2 \sum_{i=1}^{n} \{y_i \operatorname{logit}(P_i) + \log(1-P_i)\}$$

where P_i are the fitted probabilities to the binary data y_i from the selected linear regression model (Collett, 1991). To compare two models for binary data, where model 1 has parameters $\beta_0, \beta_1, \ldots \beta_h$ and deviance $D1$ and model 2 has parameters $\beta_0, \beta_1, \ldots \beta_k$ and deviance $D2$ ($k > h$) (Richards *pers.comm.*, 1996) it is necessary to know the maximised likelihood for both models, denoted by L_{c1} and L_{c2} (Collett, 1991). The difference between the deviance for each fitted model is called the Partial Deviance and is used to establish whether some predictor variables can be dropped from the model. It can be shown that Partial Deviance is $D1 - D2 = -2 [\log L_{c1} - \log L_{c2}]$ (Collett, 1991) and that the Partial Deviance follows a χ^2 distribution with degrees of freedom given by the difference in the number of parameters between two models (k-h). For a fuller explanation of deviance see Collett (1991). The result enables one to decide whether the inclusion of extra factors in a model significantly improve the fit by comparing the differences in the deviances with the relative extra percentage points of the χ^2 distribution. The calculation of the partial deviance is straightforward since included in the output of a linear regression analysis is -2 log (likelihood) for the current model being fitted (Richards *pers.comm.*, 1996).

5.2.4.5 Variable Selection

The main thrust of the work was to establish accurate estimates of the odds ratios of the disease outcome versus the exposure factors, in the presence of potential confounding and interaction effects. The first stage of the analysis was to identify the main exposure factors and the significant confounding variables. Confounding variables were selected on the basis of epidemiological considerations and previous studies, described above. However, in some circumstances statistical arguments based on the deviance may be needed to aid ones choice.

Potential confounding variables are added to the linear logistic model that contains a constant term, to assess the effect on the probability of disease. Variables which produce a significant change in deviance are included in the model. However, there are grounds to include factors which do not produce significant changes on their own, but may be included on biological grounds for further development of the model. It is possible that in combination with other variables the joint effect may result in a significant change in deviance (e.g. interaction effects). This iterative process enables the selection of the set of most parsimonious variables which may have a confounding or interactive effect to be included in the model, such that the exposure factors will be adjusted for.

The second stage is to add the exposure variables to the model, both on their own and in combination with each other, to identify the most important exposure factors. In this investigation exposure to seawater at Whitmore Bay was the first exposure factor. The second set of exposure factors were the bacterial indicator organisms *E.coli* and faecal streptococci. The significance of an exposure variable can be tested by comparing the relative change in deviance with the corresponding percentage point of the relevant χ^2 distribution. To investigate if there is any interaction between confounding variables and exposure variables, the effect on the deviance must be measured by adding such terms to the model.

5.2.4.6 Interpretation of Parameters

The coefficients of the explanatory variables in the logistic regression model are related to the odds ratio of disease associated with that variable, whilst controlling for other confounding/interaction variables. This is particularly useful when considering aetiological studies as the relative risk of disease and corresponding standard errors can be obtained from a fitted model.

In a simple model (model 1) which includes a single dichotomous exposure factor x_1 (e.g. exposure to seawater at Whitmore Bay), where $x_1 = 0$ corresponds to un-exposed and $x_1 = 1$ corresponds to exposed, the linear logistic model may be written as

$$\text{logit}\,(P_i) = \alpha + \beta_1 x_{1i}$$

Taking the exponential of the coefficient β_1 will give the odds ratio of disease for an exposed person to an unexposed person. The statistical packages used also output the standard deviation of β_1 thus enabling computation of the confidence interval for odds ratios to be obtained.

In the polychotomous model using five levels of indicator, for example *E.coli*, 4 dummy variables x_1, $x_{2,}$ x_3 and $x_{4\text{-}}$, were defined with x_2 = 0, x_3 = 0 and x_4 = 0 corresponding to the first level or reference level. The model was then expressed as:

$$\text{logit } (P_i) = \beta_0 + \beta_2 x_{2i} + \beta_3 x_{3i}, + \beta_4 x_{4i}$$

The coefficients β_2, β_3 and β_4 can be interpreted as log (odds-ratios) for individual exposure levels for the bacterial indicator organism *E.coli* relative to the first exposure level. Again the statistical software produced standard errors of the estimates which enabled computation of the confidence intervals for the odds ratios.

If a model contains both confounding and exposure factors then the parameter estimates for the exposure factor represent the log odds ratio adjusted for the confounding variables. Therefore, the linear logistic model enables the true risk attributable to the exposure variables to be obtained.

The process is more complex if interactions between an exposure factor and confounding variable are observed in the model. The estimated log odds ratios for the exposure factor will depend on the level of the confounding variable. The log odds ratio depends on the parameters associated with the exposure factor, the interaction term and the level of the confounding variable. In addition the variance of the log odds ratio depends on the variance and covariance of the parameters and the level of the confounding factor (see Hosmer and Lemeshow 1989; Collett, 1991).

5.3. Methods for Measuring Coastal Litter and Beach User Perception of Beach Debris at Whitmore Bay

5.3.1. Litter Grid Analysis

5.3.1.1. Insitu

A novel design measuring perception and tolerance of beach users to generic types of litter was employed at Whitmore Bay (Williams and Nelson, 1997a). Although Whitmore Bay is a big beach, access to large areas for field work was difficult due to high visitor loads during the hot summer days of 1995. Permission was granted by the Vale of Glamorgan Borough Council to utilise a small area to the west of the beach, limiting the study. Permission was obtained to distribute debris on the condition that the beach was cleared at the conclusion of the experiments.

To measure the perception of visitors to debris on the beach, two generic types of debris were categorised.

- General debris (Category A), which included aluminium drink cans, netting, plastic bottles and food wrapping
- Sewage related debris (Category B), which included condoms, sanitary towel plastic backing strips, plastic replica dog faeces and toilet paper

A third category C was created, comprising a mixture of groups A and B. Three sets of five grids were set up on the beach, $1.5m^2$ in size separated by a distance of 1m (Fig 5.3.1). This allowed respondents visual space to view each grid independently without their peripheral vision being distorted by information from the other grids. Each grid set comprised one category of debris, A - general debris; B - sewage related debris; C - a mixture of the two.

The grids in each set were marked one to five, with debris being deposited in increasing quantities between the grids using a linear scale. Respondents were given a questionnaire and asked to walk down the line of each set, and on the five point grid scale, select and record the grid in each category which represented a density of debris that was visually obtrusive to the extent that if the beach as a whole contained a similar density, it would be enough to deter them from a future visit. Plates 5.3.1 (Category C, grid 5) and 5.3.2 (Category A, grid 4) provide examples of two of the litter grids from the Mixed and General categories. Additional sections to the questionnaire, included questions on the main reasons for visiting Barry Island, attributes such as facilities and good water quality requiring an order of priority and socio demographic data.

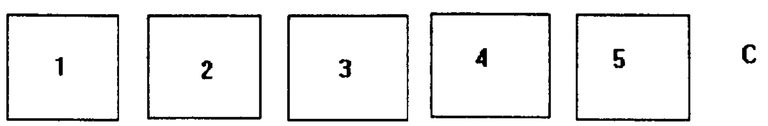

Figure 5.1 Debris Survey Grids. (A,B,C - see text)

Photographs Plate 5.3.1 and 5.3.2

Page left

5.3.1.2. Photographic Study

The use of photographic plates or slides to investigate perception of user types to specific environments have been successfully employed by various researchers (Coughlin, 1976; Dinius, 1981; Hertzgog, 1985; Williams and Lavelle, 1990; House and Herring, 1995). This technique subjects the observer to the same conditions and stimuli (Coughlin, 1976). A major problem incurred through *insitu* experiments in the field are changing environmental conditions, such as light. However, there is argument which states that using a laboratory, maintaining a controlled environment for running the tests does not capture the full experience of a site visit. Another problem highlighted in the literature is the opinion that 2-D visual stimuli is not an acceptable surrogate for landscape (Turner, 1977). Photographic plates were used in this study to investigate if a group of experts, under laboratory conditions were representative of beach users sampled in the field. In this research experts refer to a class of final year undergraduate Environmental Pollution Scientists studying coastal processes at the University of Glamorgan.

Photographs were taken of the three sets of litter grids. The photographic plates were used to assimilate the approach taken in the field. The three litter categories, general litter, sewage-related debris and a composition of the two were each represented by five photographs, laid out on laboratory benches. Students were asked to walk along each bench, corresponding to one of the litter categories and record on the five point scale which level indicated a density of litter that became visually obtrusive to the extent that it would affect their decision to make a future visit to the beach.

5.3.2. Measuring Beach Litter

Various methodologies have been developed for measuring beach litter such as the Garber Index (Garber, 1960), the Marine Litter Research Programme (TBG,1991), the Norwich Union Coastwatch project (Rees and Pond, 1994), the Environment Agency General Quality Assessment (Everard, 1995), the Thamesclean Project (Lloyd, 1996) and the MCS Beachwatch (Pollard, 1996b). Each system has been developed with a predefined agenda, making comparison between methods difficult. A recent protocol has been designed by a group of experts, the National Aquatic Litter Group (NALG), drawn from a wide range of sectors, including the Environment Agency, local authorities, academics, TBG, water industry and non-governmental organisations (NGOs; Earll and Jowett, 1998). The resultant model is being piloted and aims to standardise beach litter methodologies (refer Chapter 4). However, at the time of this survey the model was unavailable. The procedure formulated by the Norwich Union Coastwatch (NUC) study (Rees and Pond, 1994), which is a well established European project to measure litter around the coastline of Member States, was used as a platform for recording litter in this study (Appendix V).

The beach at Whitmore Bay is cleansed using a mechanical rake early each morning. To gauge the volume of litter deposited each day by visitors to the beach during the survey days, litter lying within a 5m transect straddling the strand line was recorded (Dixon and Dixon, 1981). In addition, debris type and volume were recorded in three $5m^2$ quadrats taken down the beach between the sea wall and the tide line. Their positions were selected by random numbers. The distance of the quadrats between the strandline and low water varied, dependent upon the tidal cycle. The debris quantity and type found in each transect was recorded. A slight bias was introduced to account for the natural tendency for visitors to choose positions slightly to the western end of the beach.

5.4 Questionnaire Design

5.4.1 Introduction

Research addressing the perception of the beach user to the beach environment and in particular coastal pollution is very limited (Coughlin, 1976; Morgan and Williams, 1995a; Williams and Nelson, 1996; Morgan, 1996). The WRc (1996b) stated that subjects in their study were blind to the quality of the sea as measured in microbiological terms. This point highlights a problem in the context of Coughlin's (1976) philosophy that man's perception of water pollution is a central consideration in measuring water pollution, highlighting the relationship between observer and phenomena observed, i.e. perception. This investigation takes this view one step further by considering not just the marine environment but the coast as a zone inclusive of beach.

As discussed, the nature of the study in hand is both multi-dimensional and inter-disciplinary, requiring information on social aspects of beach users and their perception to coastal pollution. There are a variety of mechanisms used to investigate user perception at recreational sites. This investigation utilised a questionnaire to obtain the social data required to analyse the less tangible components of the work. The questionnaire acted as a tool to obtain specific data on beach users' perception to beach pollution and sea pollution, provide information on the level of awareness and understanding with regard to seaside award schemes and also provide data on the activities and health of the participants, required for the epidemiological-microbiological analysis.

5.4.2 Questionnaire Design

There are two main types of questionnaire each with distinct pros and cons (Scottish Natural Heritage, 1989):

i. a self administered questionnaire, which the participant is responsible for completing themselves
ii. an 'interviewer' questionnaire administered by the interviewer.

The self administered questionnaire has the advantages of economy, speed, lack of interviewer bias and possibility of anonymity and privacy which lends itself to more candid responses (Babbie, 1979). In addition this style of questionnaire requires low intensity of labour to distribute in large numbers, which was the main reason for selecting this type for the beach survey work. The major disadvantages with which the 'interviewer' questionnaire overcomes are incomplete questionnaires and questions and mis-understood questions (Babbie, 1979). This second style of questionnaire also has a higher return rate, which made it suitable for adaptation to the post beach telephone survey and also provides the interviewer the opportunity to probe answers.

The basic framework of the beach survey questionnaires is similar to the extended questionnaire outlined in 'Methods and Techniques for Conducting Visitor Surveys (Edinburgh University, 1990). For more specific details of questions see below. All questionnaires used are displayed in Appendix II. In the design of all questions incorporated within the questionnaires used in this study three important aspects were considered. Firstly, the questions had to be pertinent to the information required. Secondly, an attempt was made to reduce any potential ambiguity within each question and thirdly an effort was made to ensure the questions were phrased in a neutral manner, not withstanding bias. This final point is made clear by Driscoll *et al.* (1994) who stated that in the investigation of perception the response can be strongly influenced by the way in which the question is asked.

Various styles of question have been detailed in survey design literature, which can be broken down into two main categories, closed and open-ended questions. Both serve specific purposes; closed questions are inclined to provide quantitative data (Alexander and Heaven, 1990; Balarajan, 1992) and open-ended questions tend to release qualitative data (Ditton and Goodale, 1973, 1974; Cutter *et al.*, 1979). A semi-structured style of questionnaire was utilised for all survey work undertaken, although predominantly structured. The questionnaire was selected to provide sufficient data for statistical analysis but also provide the opportunity for the subjects to express their own opinions. The selection of questioning techniques were chosen on their ability to be statistically analysed. The majority of questions were pre-coded (Babbie, 1979). This style of question is precise and easily transcribed, for example 'what activities have you done today?':

1. Sunbathe
2. Swim
3. Wade
4. Surf.

To gauge the participants reaction towards a particular statement attitude scales were used (Scottish Natural Heritage, 1989). For example, 'how important is the influence of a beach award flag in your choice of beach?':

1. Important
2. Vaguely important
3. Not important
4. Undecided

Ranking was used to examine the relative importance of a list of attributes by requesting the respondent to score them in order of preference (Edinburgh University, 1990). For example 'please put in order the most important reasons for selecting a beach on a scale of 1 to 5. One being the most important':

1. Beach award flag
2. Facilities
3. Clean water
4. Clean sand
5. Distance travelled to beach

Semantic differentials were used to rate an item, indicating how well the statement describes the item (Scottish Natural Heritage, 1989). For example 'how would describe the water quality on this beach?':

	very clean							very dirty
Water quality	1	2	3	4	5	6	7	8	9

5.4.3 Opportunistic Prospective Study (WHO/UNEP Protocol)

The WHO/UNEP (1989b) developed an epidemiological-microbiological protocol based on the prospective design pioneered by Cabelli (1983). The protocol formulates guidelines for health risk analysis focused on clinically controlled trials. This approach is expensive, so the WHO/UNEP (1993) re-assessed the guidelines and formed a study design for local low-cost surveillance on health risks associated with recreational waters. This strategy, termed a prospective 'opportunistic' cohort study is also aimed at small scale surveys where the expected number are low (WHO/UNEP, 1993; WHO, 1994a; WHO, 1994c). The strategy is very comprehensive, providing a questionnaire template for beach survey work, and has been used as the basis for this investigation.

For continuity the selection of the prospective 'opportunistic' cohort design has been explained in the Results and Discussion chapter, Section 6.b.1. Pruss (1996) defined cohorts as disease free populations of bathers and non bathers and the term prospective relates to the follow up survey investigating the differential in illness rates between cohorts. The cohorts can be stratified dependent on levels exposure, such as participants who waded as opposed to those that swam with head immersion. A major advantage of this style is the activity of the participant is self selected and of their own volition, allowing the study of children, which in controlled clinical trials is not acceptable for ethical reasons. To overcome the lack of medical evidence provided by the prospective cohort design, the WHO suggested that self-reporting symptoms should be validated by asking about whether incurred disease required prescriptions, medication or visits to the doctor (WHO/UNEP, 1991; WHO/UNEP, 1993). Major studies which have used this style of study include Cabelli *et al.* (1982), Brown *et al.* (1987), Lightfoot (1989), Alexander and Heaven (1990) and Balarajan (1992, 1993).

5.4.3.1 Telephone questionnaire

To determine the health risk from bathing at Whitmore Bay (1995) a post beach survey interview was required to investigate the differential in illness rates between cases and controls. The two main approaches used to obtain post survey information are a prepaid postal survey and telephone survey. Dillman (1978) stated that his research proved postal response to be effective, however, the use of a telephone interview further increased the return by 17%. Although small scale postal surveys have shown to be reasonably successful (Phillip *et al.,* 1985) in general telephone interviews are more successful (Cabelli, 1983; Nelson, 1994). In addition the WHO (1993) guidelines for epidemiological-microbiological studies promotes the use of a telephone interview in opposition to a self addressed envelope given to participants, explaining that the telephone is more effective to use and produces a higher response rate than the latter. Further advantages of a telephone interview are the potential to explain any confusion expressed by the respondent and the ability to probe for answers (Dillman, 1978).

For the reasons stated above a telephone questionnaire survey was utilised in this study (Appendix II) in preference to a postal survey. Telephone numbers were requested during the beach interviews, similar to the approach taken by Alexander and Heaven (1990). The post survey required the respondent to comment on whether they had suffered any illness since the interview day, whether they had entered the sea since that date, whether they had eaten any specified risk foods and also requested information on the health of their resident family members. The survey was conducted 10 days after the beach interview, allowing sufficient time for most waterborne pathogenic micro-organisms to incubate (Cabelli, 1983). Ten days was the average time used by most research investigations into health effects from bathing in marine waters, ranging from 7-14 days (Cabelli, 1981; Brown *et al.,* 1987; Alexander and Heaven, 1990; Balarajan, *et al.,* 1991; WHO/UNEP, 1993).

5.4.4 Questions

The WHO/UNEP (1991) suggested the use of multiple linear logistic regression to control for confounding factors and interaction effects. Examination of reports by Alexander and Heaven (1990), Jones *et al.,* (1993), Balarajan (1992, 1993) and the WRc (1996a) provided a platform to include questions based on potential confounding factors, or non-water related factors such as age, sex, visitor type, socio-economic status, previous exposure to water and specific foods eaten. Health related questions were derived from Jones *et al.,* 1993 and Balarajan, (1992, 1993).

Questions pertaining to aesthetics and beach user perception were derived from a wide range of research reports. The following water pollutants were noted as visually offensive:

- Murky water and floating objects (David, 1971; Nicolson and Mace, 1974; House and Herring, 1995; Smith *et al.,* 1995a, 1995b).
- Impaired colour and turbidity (David, 1971; Moser, 1984; Robens Institute, 1987; Burrows and House, 1989; Green and Birchmore, 1993).
- Perceived poor water quality (Ditton and Goodale, 1974; Dinius, 1981; Hertzgog, 1985; Phillip, 1990; House and Sangster, 1991; WHO, 1994a;).
- Oil (Young *et al.,* 1996; Morgan Williams, 1995a).

Questions pertaining to beach litter were derived from results of the Norwich Union Coastwatch Study (Rees and Pond, 1994), the Marine Litter Programme (TBG, 1991), the Environment Agency General Quality Assessment (Everard, 1995) and the MCS Beachwatch (Pollard, 1996b). Work carried out on female susceptibility to beach pollution was supported from reference to Williams *et al.* (1993) and Simmons and Williams (1994).

Question construction for investigation into beach users' perception of seaside award schemes was made difficult due to the lack of research in this field. Work by House and Herring (1995) and Morgan and Williams (1995b) were used as reference material. The questions were ordered such that specific names of flag systems were mentioned last to prevent predisposing the interviewee to the different types of systems available.

5.4.5 Strategy

No clear guidance exists on whether professionals or volunteers should be used in distributing questionnaires (Faris and Hart, 1995). The necessity to obtain a large survey sample, especially for the epidemiological-microbiological work in 1995 and low budget of the research programme made meant that volunteers had to be utilised. This approach was also adopted by the Robens Institute (1987). The self-administered questionnaires eliminated potential interviewer bias and meant the volunteers *did not* require intensive training (Scottish Natural Heritage, 1989). The only one-to-one contact required by the interviewers were the visual investigation of flags and sewage-related debris in the 1996 survey (see below) and the telephone survey (see above).

The volunteers were given clear instructions on distributing the questionnaire. The purpose of using a friendly and relaxed manner in addressing the subjects was stressed and the need to introduce oneself, explaining the objectives of the study and the association to the University of Glamorgan and University of Wales Institute Cardiff (WHO/UNEP, 1993). Volunteers working on the post telephone survey also received intensive training (Dillman, 1978). Before actually working on the beach surveys all volunteers were familiarised with the questionnaire and its objectives. It was believed that appeal to altruism would produce a positive feedback. At the end of each day survey notes were recorded on weather conditions, position of the tide, air and water temperature, visitor loads, visual pollution levels and unusual occurrences.

5.4.6 Surveys

No literature was evident suggesting what constituted a representative sample of beach users for the epidemiological-microbiological survey. The WHO/UNEP (1993) stated the need to secure enough sample units for both cases and controls to be statistically significant, including consideration of reduction in numbers following stratification of the sample into activity levels and non-water related factors. The aim of the 1995 survey was to was to achieve in excess of 1000 participants to adjust for the effect of stratification and account for a certain percentage of subjects who were reluctant to offer a contact number for the follow up survey. The sample size was designed from analysis of studies by Phillip (1985), Alexander and Heaven (1990), Jones *et al.* (1993). Whitmore Bay was chosen to provide sufficient number of beach users, being a large and popular resort beach. Due to limitations of the laboratory capacity to facilitate microbiological analysis on weekends the 1995 survey was based only on weekdays. This was altered for the 1996 work which was conducted over both weekdays and weekends. The sample size for the 1996 work was to obtain minimum 100 questionnaires per beach (Babbie, 1979), which included Whitmore Bay, Langland Bay and Cefn Sidan. One main criteria for selecting Langland Bay and Cefn Sidan was to achieve sufficient survey data, both beaches receive heavy visitor loads during the summer months. The recommended number of survey days by Edinburgh University (1990) is eight. This was achieved during the 1996 survey. However, the survey days were kept to a minimum during the 1995 survey, dependent upon the least time to reach 1000 respondents. The reasoning behind this was to keep the number of microbiological readings to a reasonable number to aid statistical analysis.

A pilot study was used prior to conducting both beach surveys for 1995 and 1996 to ensure that they ran smoothly, were easily understood and provided the required information. Both pilots were helpful allowing final tuning before operationalisation. The pilot studies highlighted two important points. Firstly, they identified a window period between 11.00am and 3.00pm which constituted the highest visitor density and also the willingness of people to be involved in beach surveys. In addition beach users proved tolerant to questionnaires taking in excess of 25 minutes to complete. The pilot questionnaires were tested on a wide spectrum of people including beach users, academics and beach lifeguards

Research suggests employing random sampling in selecting participants for recreational studies, to reduce bias (Babbie, 1979). However, although sound in theory, with the requirement to obtain such a large sample, especially during the 1995 work and the dynamic nature of people on the beach (Morgan, 1996) a systematic approach was utilised.

Three questionnaires were designed for the 1995 survey (Appendix II). QA addressed respondents aged over 10 and QB was split into two sections. Section one addressed the health and activities of children 10 and under, to be filled in by the parent. Section two addressed the perception to coastal pollution and seaside award schemes of the parents. Age 10 was used as a cut-off point discriminating between Secondary and Comprehensive education. Both QA and QB were designed to take between 20-25 minutes. The third questionnaire was the telephone interview schedule which took approximately 5 minutes to run. Only one questionnaire was required for the 1996 survey which concentrated on developing the perception work of the 1995 survey further, at an additional two beaches, Langland Bay and Cefn Sidan, dropping the epidemiological-microbiological work. The 1996 survey was operationalised in conjunction with visual stimuli (photographs) of coastal pollution items and seaside awards (Appendix II). Levels of exposure to beach contact were derived from the WHO/UNEP (19936) questionnaire. The questionnaire was designed to take between 10-15 minutes.

5.4.7 Data Analysis

Questionnaires for both the 1995 and 1996 surveys followed a similar template for comparison of results. The questionnaires were coded and input into statistical packages which included SPSS (1995), Jandel Scientific (1995), Statistica (Statsoft, 1993) and the spreadsheet Excel (Windows, 1996). There was a large potential for combinations of variables, but only the most pertinent ones were selected for analysis. All analysis was carried at the $P=0.05$ level unless otherwise stated. The Kolmogorov-Smirnov Normality test was used to test whether the data followed a normal distribution (Siegel, 1956). Non-parametric tests including the Kruskal-Wallis Analysis of Variance on Ranks (Analysis of Variance), Mann Whitney Rank Sum Test and χ^2 analysis were employed appropriately to data that *did not* follow normal distributions. The tests used are referenced throughout the text. More specific statistical testing is detailed in the respective methods sections.

5.4.8 Weaknesses

Recreational survey work using questionnaires carry inherent weaknesses. For a comprehensive account of these weaknesses consult Babbie (1979) and Edinburgh University (1990).

- The large volume of questionnaires required for both the 1995 and 1996 surveys meant that it was necessary to employ 'self-administered' questionnaires. Although there are advantages of using this approach over an 'interviewer' style questionnaire it did lead to a small tendency for some respondents to leave gaps.

- On-site questionnaires may prompt the participant to perceive aspects of the environment that they may not have previously observed, confounding the results.

- Variations in response of the on-site questionnaires may be due to the ever changing dynamic conditions of the coastal environment.

- Respondents involved in the epidemiological-microbiological survey investigating health risks may have produced biased answers in self-reporting their own symptoms, due to their awareness of the study purpose. This is a common problem with epidemiological studies.

- A series of questions incorporated on the 1995 questionnaire requested the participants to select the three most important aspects from a list. This style of question does not lend itself to statistical analysis. The problem was overcome in the design of the 1996 survey by requesting respondents to rank lists of attributes.

- Random sampling is a technique which removes study bias. However, the need for large survey numbers, limited financial and human resources and the dynamic movement of people on the beach necessitated the use of a systematic approach to sampling.

Chapter 6(a) Results and Discussion

6.a WATER QUALITY

6.a.1 Microbiological Quality of Barry Island Bathing Water

The main thrust of the research with respect to water quality monitoring was to investigate if a dose response relationship between bacterial concentration and incidence of water-related illness existed at Whitmore Bay (refer Section 6.b.1). The second objective was to examine the water quality over the survey periods 1995 and 1996, observing temporal, spatial and tidal fluctuations. Three indicators were monitored during 1995 and 1996. *E.coli* and faecal streptococci were tested for during both surveys in line with proposals to the European Bathing Water Directive (CEC, 1997). F-specific RNA phages were tested for during 1995 to accommodate potential future inclusion of bacteriophages into the Directive (Nelson and Williams, 1997). Table (6.a.1) summarises the main bacteriological determinands in the Bathing Water Directive, which is listed in full in Appendix IV.

No water sampling was done at either Langland Bay or Cefn Sidan. Inspection of the results produced by the Environment Agency show that Langland has had a 75% pass rate with EC Mandatory standards since 1986 (Environment Agency, 1997). Cefn Sidan has had excellent water quality results, having a 100% compliance rate with EC Mandatory standards, but also meeting Guideline standards for faecal streptococci (Environment Agency, 1997). This has enabled it to receive the EC Blue Flag (FEEE, 1997) 9 times in the past including the 1997 bathing season (refer Section 2.4).

EC Com(97) 585 Final Amendments	**E.coli 100ml^{-1}**	**F.streps 100ml^{-1}**	**Bacteriophages**
Imperative (Mandatory) level 95% of samples should not exceed this figure	2000	100	No value
Guide level 80% of samples should not exceed this figure	100	50	No value

Table 6.a.1 Summary of Bacterial Indicators in EC Bathing Water Directive

Whitmore Bay is an identified beach, which means the water quality must be compliant with criteria set by the EC Bathing Water Directive (CEC, 1976a). The Environment Agency are obliged to sample identified recreational waters once per week during the bathing season commencing May 15 to September 30 (Environment Agency, 1996). Selection of one sample from one sampling point per week is believed to be inadequate providing only a snapshot of the water quality, not taking into account environmental and physical conditions over time and space (Rees *pers.comm.,* 1995; Fleisher, 1990b). The water sampling programme used in this study was more intensive than that carried out for example by Alexander and Heaven (1990) who sampled once per day at two sites and Lightfoot (1989) who took two samples per day at each location.

Historically, Whitmore Bay has a poor record of compliance with EC bathing water standards, which has prevented it from being eligible for a European Blue Flag. Since 1986, the Bay has only achieved five passes in 1991, 1993, and 1995-1997 (Environment Agency, 1997), a success of rate of only 42%. Although the previous three years have resulted in a pass, which appears promising, compliance has been based on the Bathing Water Directive (CEC, 1976a) set in 1976. New reforms to the Directive (1997), which are yet to be enforced, will undoubtedly affect the ability of Whitmore Bay to reach new standards, especially that of faecal streptococci (Table 6.a.1). Based on the new Mandatory standard for faecal streptococci, Whitmore Bay would have failed every year between 1991-1997, with a pass rate averaging only 18% of all annual samples (NRA, 1991-1995; Environment Agency, 1996-1997).

Two sewage outfalls are discharged into waters off Barry and prevailing winds are south westerly (refer Section 2.1). Water sampling points S1 were central Whitmore Bay and S2 300m west of S1, detailed on Figure 2.1. For a full description over selection of sampling points see section 6.1.1.1.

6.a.2 Survey 1995

The water analysis for 1995 covered 6 survey days, sampling at two sites at three points in time, 11.00am, 1.00pm and 3.00pm. Due to time restrictions a wider spatial distribution of samples was taken in preference to replicate sampling. For each site three samples were taken daily for *E.coli*, faecal streptococci and F specific RNA bacteriophages, yielding a total of 18 samples over the sampling period, per determinand. The F specific RNA bacteriophages were tested at acer Laboratories, Bridgend, South Wales. The complete data set for *E.coli* and faecal streptococci over the 1995 water quality analysis, displaying all dilutions across sample sites one and two is listed in Appendix III, summarised in Table 6.a.2.

All analyses were conducted using Jandel Scientific (1995) statistical software. The geometric mean was selected over the arithmetic to rationalise the data for use in the statistical analysis. This method reduces the effect of outliers and is routinely used in epidemiological-microbiological studies (Cabelli, 1983; Balarajan *et al.*, 1991; Jones *et al.*, 1993). The arithmetic mean has been computed and also the range for comparison with the Environment Agency data.

		E.coli Per 100ml					F.Strp Per 100ml			
Date		Arith.	Geo.	Range			Arith.	Geo.	Range	
Aug.95		Mean	Mean	Min.	Max.		Mean	Mean	Min.	Max.
Mon. 7	**S1**	26667	21918	9000	45000		1010	725	190	1540
	S2	14800	14741	13000	16000		348	290	34	124
Daily av.		20734	17975	9000	45000		679	459	34	1540
Tues. 8	**S1**	8933	4816	1500	21900		373	341	210	590
	S2	3390	2857	1470	6100		763	267	91	2100
Daily av.		6162	3709	1470	21900		568	301	91	2100
Wed.9	**S1**	3350	2158	1800	3100		263	261	220	290
	S2	1810	1500	730	3300		308	258	103	450
Daily av.		2580	1799	730	3300		285	260	103	450
Wk Av.		9825	4832	730	45000		511	330	34	2100
Mon. 14	**S1**	6967	5206	3000	14700		6937	1048	162	20300
	S2	5867	5825	5100	6800		4510	2121	430	11100
Daily av.		6417	5507	3000	14700		5723	1491	162	11100
NRA 14		***1170***					***840***			
Tues. 15	**S1**	3007	2806	1820	4900		539	364	118	1140
	S2	1927	1815	1330	2900		523	457	130	930
Daily av.		2467	2257	1330	4900		531	408	118	1140
Wed.16	**S1**	3007	1193	1820	4900		539	308	118	1140
	S2	1927	928	1330	2900		523	372	310	930
Daily av.		2467	1052	1330	4900		531	339	114	118
Wk Av.		3784	2356	1330	14700		2262	591	118	20300
Svy. Av.		**6805**	**3374**	**730**	**14700**		**1387**	**442**	**34**	**20300**
NRA Av.		**699**	**600**	**160**	**2200**		**307**	**213**	**81**	**1200**

Table 6.a.2 Summary Water Quality Results 1995 (Source NRA, 1996)

6.a.2.1 Water Quality Analysis

Counts for both *E.coli* and faecal streptococci averaged 3374/100ml and 442/100ml respectively (Nelson and Williams, 1997). Maximum counts for the coliform bacteria reached 45,000/100ml and the faecal streptococci count reached 20,300/100ml (Nelson and Williams, 1997). These figures well exceed the criteria set by the EC bathing water directive, the Mandatory level for *E.coli* being 2000/100ml with a Guideline of 100/100ml and the Mandatory standard faecal streptococci 100/100ml with a Guideline of 50/100ml. Only 33% of the daily samples met the *E.coli* Mandatory level and no faecal streptococci daily counts met the Mandatory standard. No colonies of F specific RNA phage were found (Nelson *et al.,* in press (a)), which might be due to analytical procedures (Rees *pers.comm.,* 1995b). Although health risk was not correlated with bacterial indicator density in this study (refer Section 6.b.1), Kay *et al.,* (1994) reported a level of faecal streptococci in excess of 32/100ml to show a significant increase in reported incidence of illness. This value is over three times less than the proposed Guideline value of faecal streptococci (CEC, 1997).

6.a.2.2 Statistical Distribution

The bacterial data sets for sample sites one and two (Table 6.a.2) failed the Kolmogorov-Smirnov normality test ($P = <0.05$), indicating that the distributions of both *E.coli* and faecal streptococci vary significantly from the pattern expected if the data was drawn from a population with a normal distribution (Jandel Scientific, 1995). Non parametric statistical analysis was applied to the data including the Mann Whitney Rank Sum Test, a powerful technique suited to detecting if samples are likely to have originated from the same parent population (Siegel, 1956; Porkess, 1988). The Pearson Product Moment Correlation was used to investigate whether a relationship between bacterial concentration over time, space and tidal variation existed.

6.a.2.3 Comparison with Environment Agency Results

The water sampling strategy for the 1995 survey was more intense than that employed by the Environment Agency (NRA, 1996). As already indicated, the Environment Agency only take one sample per week from one central location on the beach. The statistics provided by the Environment Agency include the arithmetic mean and the range. The only directly comparable result was on 14 August, 1995 which was one of the sample days for this study and when the Environment Agency also tested. Results of this study revealed the arithmetic mean value for *E.coli* to be 5.5 times higher than the Environment Agency arithmetic mean value for *E.coli*. The arithmetic mean value for faecal streptococci was also higher in this study than the Environment Agency value, of the order 6.8 times. For the purpose of this study the annual geometric mean value taking into account all Environment Agency results for the bathing season 1995 was calculated, which was provided in raw data format (NRA, 1996). This value was compared with the geometric mean value from all results taken during the survey 1995. Table 6.a.2 shows the geometric mean value recorded in this study for *E.coli* to be 5.6 times higher than the Environment Agency geometric mean value. Similarly, the geometric mean recorded in this study for faecal streptococci was over 2 times larger than the corresponding Environment Agency geometric mean value (NRA, 1996). One explanation for higher counts recorded in this study is that the water analysis was carried out more rapidly than samples taken by the Environment Agency who have several beaches to monitor compared to just one. Therefore, these samples were resuscitated much sooner than Environment Agency samples. Out of the 20 samples taken by the Environment Agency, 19 of the *E.coli* passed the Mandatory standard of 2000 per 100 ml, but only 2 samples of faecal streptococci were under the Guideline value of 100 counts per 100 ml (CEC, 1976a). This enabled Whitmore Bay to gain an overall pass. However, as already explained the new Mandatory standard for the reformed EC Bathing Water Directive is 100/100 ml requiring a 95% compliance rate, which would constitute a fail.

6.a.2.4 Temporal Variation

Figure 6.a.1 shows geometric mean levels of *E.coli* and faecal streptococci counts to vary over the survey days during August (Nelson and Williams, 1997). Higher levels were recorded following the weekend. Monday 7th *E.coli* was 17 975/100ml and faecal streptococci were 459/100ml and on Monday 14th *E.coli* was 5507/100ml and faecal streptococci were 330/100ml. There was a general trend for bacteria levels to drop off towards the end of each week. Although these observations were only over a 2 week period, higher concentrations of bacteria after the weekend may be due to increased visitor loads on Saturday and Sundays.

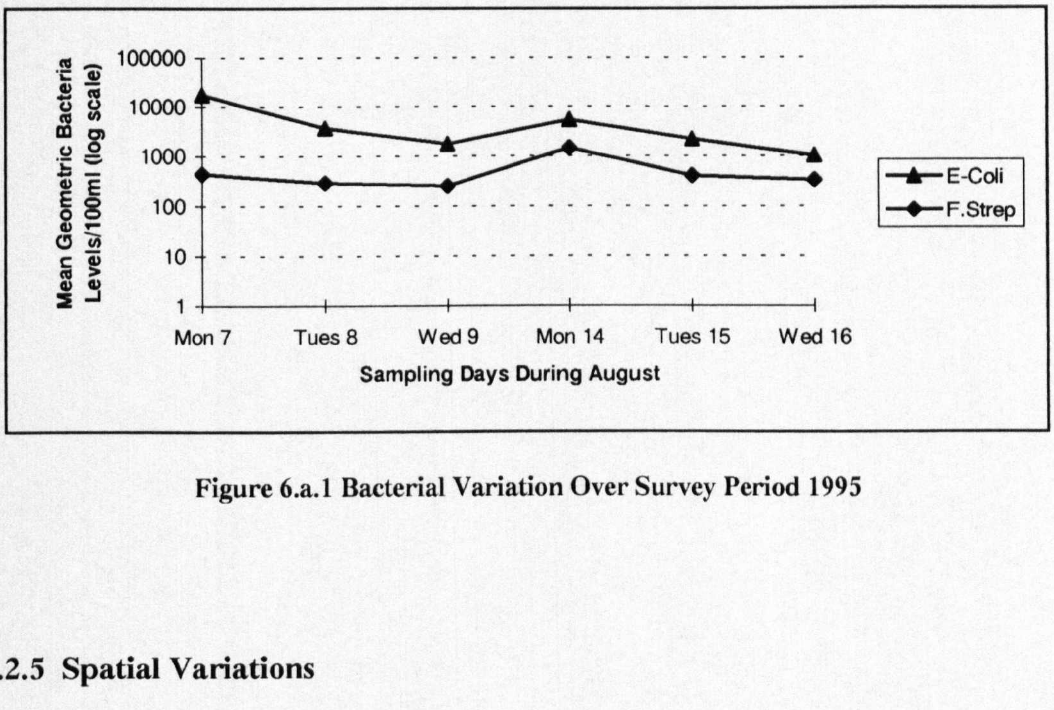

Figure 6.a.1 Bacterial Variation Over Survey Period 1995

6.a.2.5 Spatial Variations

Figures 6.a.2 and 6.a.3 show the variation between sample sites 1 and 2 for both *E.coli* and faecal streptococci respectively (Nelson *et al.,* in press (a)). Both sites appear similar for *E.coli* and faecal streptococci supported by the Mann Whitney Rank Sum Test which was applied to the data. In each case the differences in the median values among the two groups were not great enough to exclude the possibility that the difference is due to random sampling variability. There was not a statistically significant difference between sampling sites at the P=0.05 level for both *E.coli* (P = 0.393) or faecal streptococci (P = 0.812; Jandel Scientific, 1995).

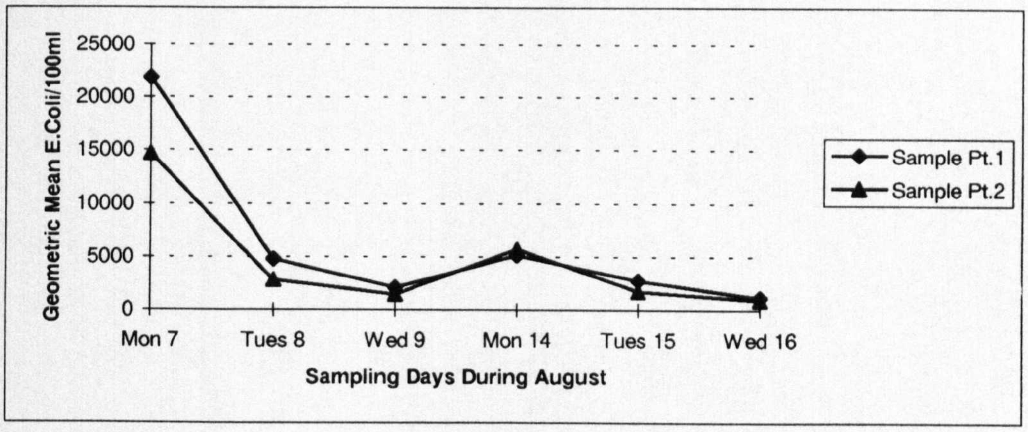

Figure 6.a.2 Comparison of *E.coli* Between Sample Sites 1995

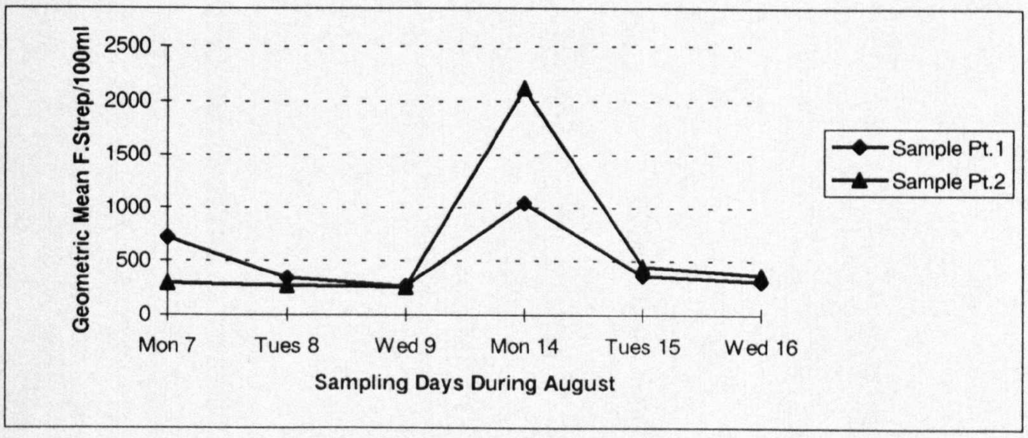

Figure 6.a.3 Comparison of Faecal Streptococci Between Sample Sites 1995

6.a.2.6 Tidal Variations

To investigate if there was a correlation between bacteria concentration and tide position, the data was sub-divided into three categories depending upon what point in the tidal cycle they were obtained. Samples taken at high tide were code 1, samples taken at medium tide were coded 2 and samples taken at Low tide were coded 3. The Pearson Product Moment Coefficient was used to determine the degree of correlation. First each sample point for *E.coli* and faecal streptococci were run against the tidal cycle, producing 18 data pairs per test and then both sample points for each indicator were pooled and run against the tidal cycle, producing 36 data pairs per test. This totalled five tests. In each case there were no significant relationships between any pair of variables in the correlation table ($P > 0.050$), indicating that these data values do not support any correlation between *E.coli* or faecal streptococci and fluctuations in the tidal cycle (Jandel Scientific, 1995).

6.a.2.7 Correlation between E.coli and Faecal Streptococci

The Pearson Product Moment Correlation was used to investigate if there was any correlation between both indicators *E.coli* and faecal streptococci. The first two tests were between sample sites 1 and 2 producing 18 data pairs. Data from both sampling points for the respective indicators were pooled and subjected to the Pearson test, producing 36 data pairs. There were no significant relationships between any pair of variables in the correlation table ($P > 0.05$), indicating there not to be a correlation between *E.coli* and faecal streptococci at Whitmore Bay over the sampling period. However, they do appear to follow the same trends, Figure 6.a.2 and 6.a.3.

6.a.3 Survey 1996

Logistical and financial constraints prohibited further health risk analysis using water quality monitoring. The 1996 survey was used to investigate the bacterial quality of the water at Whitmore Bay and verify the sampling technique by using replicate sampling. In addition comparison with the Environment Agency results was made. Samples were taken over three days on the 2, 3 and 5 of September 1996. The central location (Figure 2.1) was used to take the three samples over three points in time, 11.00am, 1.00pm and 3.00pm. Similar to the 1995 data, the geometric mean and range were used to statistically analyse the data; the arithmetic mean has also been calculated. The full set of results including dilutions can be viewed in Appendix III, which shows that the replicates were closely matched. Table 6.a.3 summarises the data showing *E.coli* to fall below the EC Mandatory standard of 2000/100ml on all three days, having an average of 1226/100ml. The data for faecal streptococci showed the average to be 170/100ml which is still 70% higher than the stipulated EC Mandatory criteria.

	E.coli	**per**	**100ml**			**F.Strp**	**per**	**100ml**	
Date Sept.96	**Arith. mean**	**Geo. Mean**	**Range Min.**	**Max.**		**Arith. mean**	**Geo. Mean**	**Range Min.**	**Max.**
Mon. 2	1562	1360	400	3000		137	141	60	290
Tues. 3	1494	1251	300	2900		254	271	110	700
Thurs.5	1342	1083	500	2600		147	129	40	270
E.A. Thurs. 5	*285*					*115*			
Survey Average	1466	1226	300	3000		179	170	40	270
E.A. Annual	424	354	73	900		194	150	38	640

Table 6.a.3 Summary Water Quality Results 1996

Source: Environment Agency Results, Environment Agency, 1997

6.a.3.1 Statistical Distribution

Due to the limited number of data points it would be unwise to attempt to determine if the bacteria followed a normal distribution. Therefore non-parametric statistical techniques were used to analyse the data including the Mann Whitney Rank Sum Test (see above). The Pearson Product Moment Correlation was used to establish if any correlations between time, space and tidal cycle against bacterial counts were evident and to ascertain if a correlation between *E.coli* and faecal streptococci existed.

6.a.3.2 Comparison with Environment Agency Results

The sampling strategy used for the 1996 survey was more intense that that used by the Environment Agency (Environment Agency, 1996b). The arithmetic mean value obtained by the Environment Agency for *E.coli* on Thursday the 5 September, 1996, was 285/100ml (Environment Agency, 1996b) which is 4.7 times lower than the corresponding value of 1342/100ml taken on the same day in this study, and from the same sampling point (Table 6.a.3). The level of faecal streptococci reported by the Environment Agency (Environment Agency, 1996b) for the same day, 5 September, 1996, was 115/100ml compared to 147/100ml found here. There was considerable discrepancy between the two *E.coli* values, which again could be down to faster recovery of indicator in this study compared to that of the Environment Agency. The faecal streptococci value is much closer, but faecal streptococci are more resistant to environment decay than *E.coli* (Kay *et al.,* 1994). For the purpose of this study the annual geometric mean value taking into account all Environment Agency results for the bathing season 1996 was computed, which was provided in raw data format (Environment Agency, 1997b). This value was compared with the geometric mean value from all results taken during the survey 1996. Table 6.a.3 shows the geometric mean value recorded in this study for *E.coli* to be 3.5 times higher than the Environment Agency geometric mean value. The geometric mean recorded in this study for faecal streptococci was very close to the corresponding Environment Agency geometric mean value, being only 10% higher (Environment Agency, 1997b).

6.a.3.3 Temporal and Spatial Variation

Although only three survey days were monitored, the levels of *E.coli* followed a similar pattern to the 1995 data dropping off towards the end of the week (Figure 6.a.4). This trend was not mirrored by faecal streptococci, which displayed the highest value mid week. Spatial variations could not be analysed as only one site was monitored.

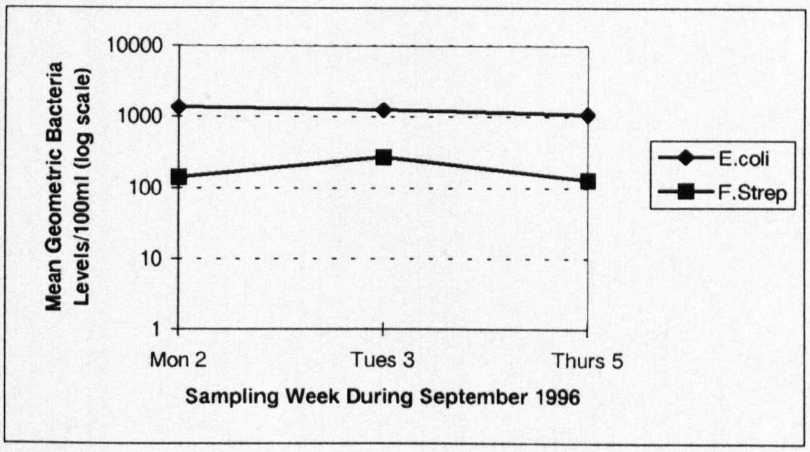

Figure 6.3.a.4 Bacterial Variation Over Survey Period 1996

6.a.3.4 Tidal Variations

In a similar manner to statistical treatment applied to the 1995 data, the 1996 data was analysed to investigate if there was a correlation between bacteria concentration and position of the tide. The tidal coding system was repeated for the 1996 data as applied to the 1995 data. The Pearson Product Moment Coefficient was used to determine the degree of correlation. *E.coli* and faecal streptococci were run against the tidal cycle, producing 9 data pairs per test. In each case there was no significant relationships between any pair of variables in the correlation table ($P > 0.050$), indicating that these data values do not support any correlation between *E.coli* and faecal streptococci and fluctuations in the tidal cycle (Jandel Scientific, 1995). Caution must be applied to this result with only limited data values being tested.

6.a.3.5 Correlation between E.coli and Faecal Streptococci

The Pearson Product Moment Correlation was applied to the data to investigate if the was any correlation between both indicators *E.coli* and faecal streptococci. Nine data pairs were analysed. There were no significant relationships between any pair of variables in the correlation table ($P > 0.050$), indicating that there was not a correlation between *E.coli* and faecal streptococci at Whitmore Bay over the sampling period (Jandel Scientific, 1995). Caution must be applied to this result with only limited data values being tested.

6.a.4 Summary of Water Quality Results

During the 1995 survey maximum counts recorded for *E.coli* reached 45 000/100ml and maximum counts of faecal streptococci reached 26 000/100ml. Geometric mean counts over the survey period for *E.coli* were 3374/100ml and for faecal streptococci were 442/100ml. During the 1996 survey maximum counts recorded for *E.coli* reached 2625/100ml and maximum counts of faecal streptococci reached 426/100ml. Geometric mean counts over the survey period for *E.coli* were 1226/100ml and for faecal streptococci were 170/100ml. No F specific RNA phages were found in any of the water samples (acer Laboratories).

The data for both surveys, 1995 and 1996 showed temporal fluctuations throughout the week, except for faecal streptococci (1996). Higher values were recorded following weekends, possibly due to high visitor loads on Saturdays and Sundays. Spatial variation analysis was only conducted for 1995 at two sampling sites, which yielded no statistically significant difference between sites for either *E.coli* or faecal streptococci. Also, no statistical variation was observed between bacterial counts and tidal positions in both surveys.

The sampling programme used in this study, sampling at two hourly intervals between 11.00am and 3.00pm inclusive was significantly more intense than the sampling programme utilised by the Environment Agency. During the 1995 survey two sampling points were analysed, central and west end of the beach. The 1996 was only conducted at the central location, but concentrated on replicate sampling.

6.a.5 Discussion of Water Quality Results

In general average levels of bacteria recorded over both bathing seasons were at least three times higher than the corresponding Environment Agency results, except for counts of faecal streptococci in 1996 (refer Tables 7.a.2 and 7.a.3). A plausible explanation for the higher bacterial density recorded in this study compared to the Environment Agency may be due to the faster processing time between sampling and incubation, reducing die off. For a direct comparison of results with the Environment Agency it would have been necessary to process the samples following the same time period between sampling and incubation. However, confined resources and limited access to the microbiology laboratory made this impractical. The main emphasis of this research was to obtain the most accurate estimation of indicator levels for use in the logistic regression modelling of health risk.

The water sampling strategy employed was significantly more intensive than that used by the Environment Agency, sampling every 2 hours between 11.00am and 3.00pm, accounting for temporal, spatial and tidal variations. The Environment Agency are only required to sample once per week at each beach from one location, in line with the EC Bathing Water Directive (CEC, 1976a). It is widely accepted that the water sampling programme stipulated in the EC Bathing Water Directive (CEC, 1976a) is inadequate (Fleisher, 1990a) and only provides a snapshot result (Nelson *et al.,* in press (a)), described by Rees as being an almost arbitrary set of statistics (Robens Institute, 1997).

To obtain a robust set of results indicative of beach water quality a minimum of 3 sampling sites per beach should be examined, using replicate sampling during the times of highest swimmer density.

Water quality results produced by the Environment Agency (1997) show Whitmore Bay to inconsistently pass Mandatory standards laid out in the EC Bathing Water Directive (1976), although it obtained a pass for both bathing seasons 1995 and 1996. Langland Bay tends to achieve higher water quality than Whitmore Bay, and achieved Mandatory standards for 1995 and 1996. Results produced by the Environment Agency (1997) for Cefn Sidan, apart from an anomaly regarding faecal streptococci in 1997, show water quality to consistently be of very high standard, frequently achieving Guideline standards (CEC, 1976a).

It has been shown that faecal streptococci is a better indicator of enteroviruses than total coliforms and *E.coli* (Kay *et al.,* 1994) and this has prompted the EC to reform the current Bathing Water Directive (1997), although it has not been implemented yet. The main revisions transform the existing Guideline standard of 100 per 100 ml for faecal streptococci to a Mandatory standard of 100 per 100 ml. In addition the total coliforms determinand has been dropped, whilst retaining the original criteria for *E.coli.* If these reforms are implemented there will undoubtedly be an increase in the number of British beaches failing to comply with EC water quality standards (Environment Agency water quality results 1985-1998).

Pro-active developments are taking place in Wales; Welsh Water have pledged to spend up to £600m over a five year period (1995-2000) as part of a coastal investment programme (Welsh Water, 1996a). This will lead to the installation of ultra-violet light disinfection at sewerage plants along the Welsh coast, which will almost surely improve the quality of bathing waters by the Millenium. Unfortunately, even with new sewerage systems in use in West Wales certain beaches within the locality are still failing EC standards due to uncontrollable agricultural run-off (Lowe, 1996).

Chapter 6(b) Results and Discussion

6.b HEALTH RISK ANALYSIS

6.b.1 Prospective Cohort Design

A detailed review of the pros and cons of the four main types of study design used in epidemiological investigations of this nature is given in section Chapter 3, Section 3.6. In brief the prospective cohort study chosen was pioneered by Cabelli and co-workers (1979, 1982) and endorsed by the WHO (1989b). The WHO/UNEP (1991) have selected three research methods for epidemiological studies: the cohort study, the controlled clinical study and the opportunisitic cohort study. The opportunistic (prospective) cohort approach using a post interview telephone survey was selected for two main reasons:

i. it relies on beach choice, activities being determined by the participants own volition, overriding ethical problems encountered with some of the other study designs, such as Kay's controlled clinical cohort work (Kay *et al.,* 1994). This allows survey work to be carried out on beaches that do not necessarily meet standards set by the EC Bathing Water Directive (CEC, 1976a), and also allows the inclusion of children in the sample group.

ii. secondly, this style is applicable for small scale low-cost surveillance studies where resources are low (WHO/UNEP, 1993; Phillip, 1994b). The main criticism of this type of investigation is the fact that it is based on self-reported symptoms.

Recommendations by the WHO/UNEP (1991) stated that self-reported symptoms were acceptable where clinical data is unavailable but should be backed up with information obtained on medical attention required. Similar studies utilising the prospective cohort design have been conducted by Cabelli *et al.,* (1982), Alexander and Heaven (1992) and more recently the health risk from bathing in sewage contaminated waters investigation commissioned by the DoE, reported by Pike (1994).

6.b.2 Distribution of Data

A total of 1276 survey responses were obtained using questionnaires A and B Whitmore Bay, 1995 (refer Section 6.d.1). A yield of 593 answered the request for their telephone number, of which 585 were successfully contacted to investigate the health risk from exposure to seawater. Ninety seven of those contacted by telephone reported experiencing symptoms within 10 days of their day at the beach. The following figures and percentages are all based on participants contacted via telephone (585), used for the epidemiological-microbiological analysis, unless otherwise stated.

6.b.2.1 Age Distribution

6.b.2.1.1 Age Against Gender

Figure 6.b.1 shows that the largest segment populating the beach were children (27%) followed closely by the 30-39 age group (23%) and then the 40-59 age group (21%). The smallest group was the >60 age range (6%), followed by the 10-19 age group (11%) and finally the 20-29 age group (12%). Figure 6.b.2 defines gender against age for the whole health risk sample. In every age range females (Figure 6.b.2) were in significantly higher numbers than males, except for under tens. In this range 54% were girls compared to 46% boys. The females in the 20-29 and 30-39 year old categories again are highly represented, making up 91% and 74% of their age groups, respectively. This adds weight to the idea that these may well be parents, of which most are likely to be mothers.

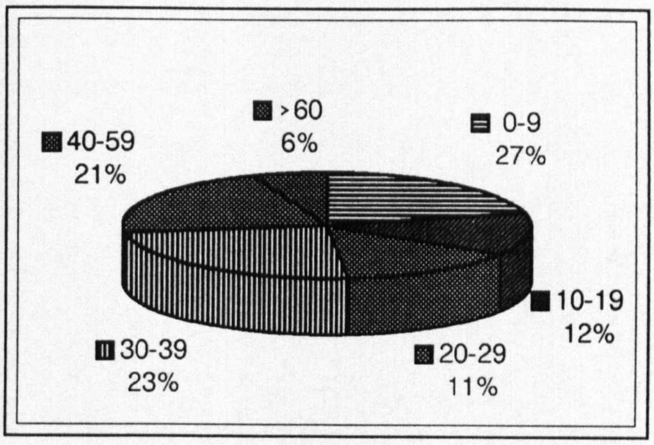

Figure 6.b.1 - Age Distribution

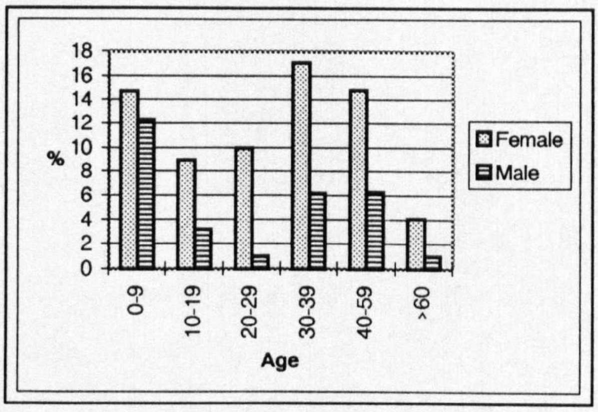

Figure 6.b.2 - Age vs Gender

6.b.2.1.2 Age Against Illness

Seventeen percent of the sample reported illness. Children under the age of ten had the highest incidence of illness, representing 7% of the total population (Figure 6.b.3.). The percentage of children under the age of 10 who were ill as a ratio of total children under the age of 10 was 28.5%, which is much higher than the ratio of children calculated as a proportion of the total population. Least likely to be ill were older aged people in the range >60 years, 0.2% of the sample and 3% of their age range. The four age categories constituting the range 10-59 produced very similar numbers of reported symptoms, *circa* 2% of the total population. However, when illness rates are expressed as a proportion of the total number in each age category, results display a much different picture. Under these calculations age group 20-29 were the second most likely to be ill, 22%, followed by 10-19 group with 15.2%. Age Groups 30-39 and 40-59 had illness rates of 9.5% and 11.% respectively. These results are not directly comparable to the raw odds ratios which used collapsed contingency tables due to data limitations, explained later.

6.b.2.1.3 Age Against Entering the Water

The results clearly show that entry into the water has a huge significance on illness rates. Ninety seven percent of the sample who reported illness had had contact with the water during the interview days. Children under the age of 10 proved to be the group most likely to have had contact with the water, 85% entered as opposed to 15% that did not enter (Figure 6.b.4). Relative to the total number within each respective age category exposed to the water, young people between 10-19 were observed to be the second most likely to enter the water, 65%, followed closely by those over 60 years with 63%. The remaining 3 age categories, 20-29, 30-39 and 40-59 had similar rates of exposure to water which were 58%, 53% and 56% respectively. It is worthwhile observing that a much higher number of children were present on the beach than any age group and that young people between 0-19 years represented almost one half of the whole sample who entered the water, 48% (Figure 6.b.4)

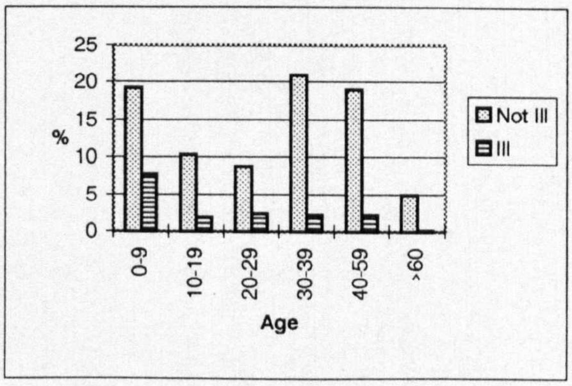

Figure 6.b.3 - Age vs Illness

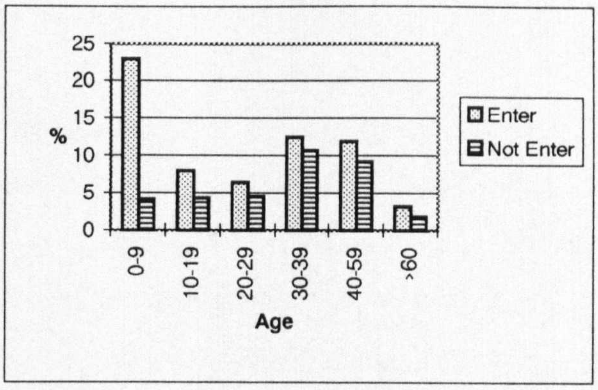

Figure 6.b.4 - Age vs Entering the Water

6.b.2.2 Socio-Economic Distribution

Socio-economic status was defined using five categories, employed, housewife, student (post 16 age groups included), unemployed and retired. Of the sample group over one third of those contacted were employed (38%), followed by housewives (31%), highlighted in Figure 6.b.5. Students represented 20% of respondents and the unemployed and retired groups made up the remaining 11%.

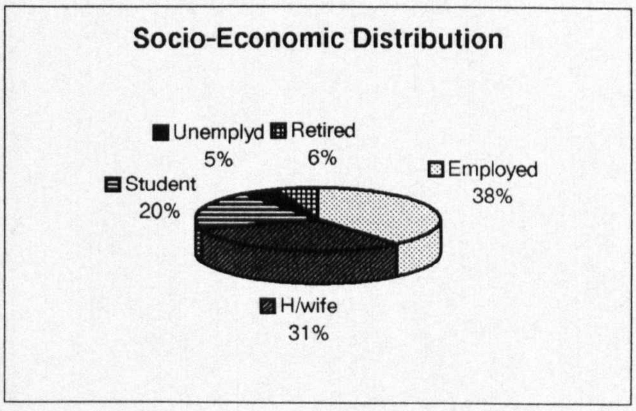

Figure 6.b.5 - Socio-Economic Status Distribution

6.b.2.2.1 Socio-Economic Status Against Illness

Fifty one percent of students reported illness, the highest incidence within the sample, represented by 10.1% of all those interviewed. This ratio is significantly more than any other group (Figure 6.b.6), which can be accounted for a by a large section of this class consisting of children aged between 0 and 9 years, who reported highest illness rates. Lowest reporting of illness were from those unemployed, 3% of their group. Employed persons and housewives displayed similar rates of illness with respect to their individual classes, 7% and 8% respectively.

6.b.2.2.2 Socio-Economic Status Against Entering the Water

Figure 6.b.7 shows the distribution of socio-economic class who entered the water. Students were also the most likely to enter the water. Again this is mainly due to children and young people between the ages of 0 and 19 comprising most of the student class, the age range most likely to enter the water (Figure 6.b.4). Of the student group 82% entered the water and 18% did not enter, compared to 54% of employed persons who entered against 46% who did not enter and 66% of housewives entered against 34% who refrained from entering.

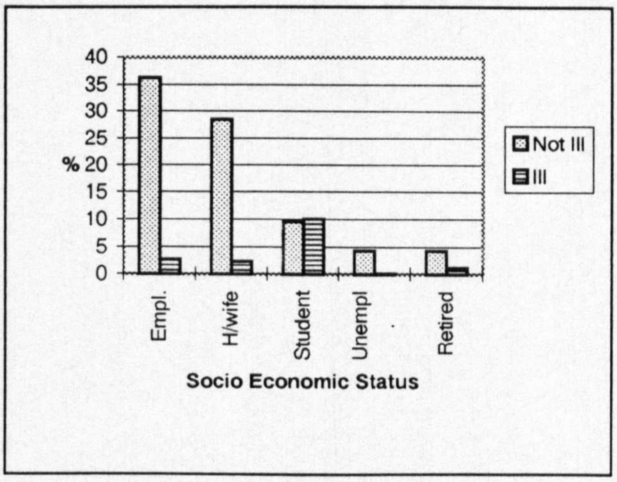

Figure 6.b.6 - Socio-Economic Status vs Illness

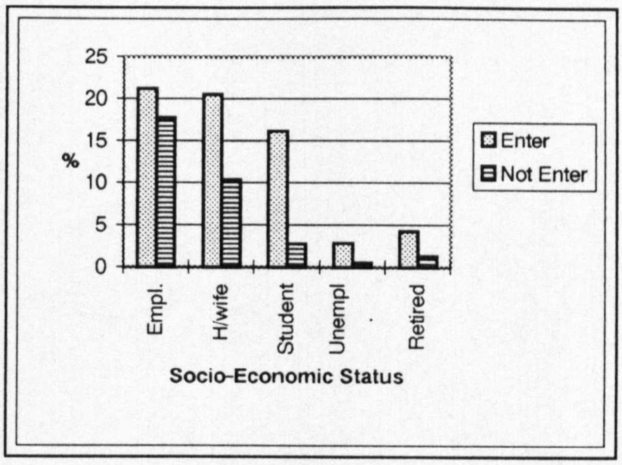

Figure 6.b.7 - Socio-Economic Status vs Entering the Water

6.b.2.3 Gender Distribution

Figures 6.b.8 and 6.b.9 show that the study was unevenly balanced across gender. Of the total number interviewed, 409 were female representing 70% of the sample population, in contrast to 176 males which accounted for 30% of the sample. Although a higher percentage of females reported illness (10% of the total sample), compared to 7% of males, in relative terms this number is much lower. Of the female class only 14% reported illness in contrast to 22% of all males. Figure 6.b.9 also highlights that the significantly higher volume of females present on the beach accounts for the higher number that entered the water. However, in relative terms males were more likely to enter the water (73%) than females (62%).

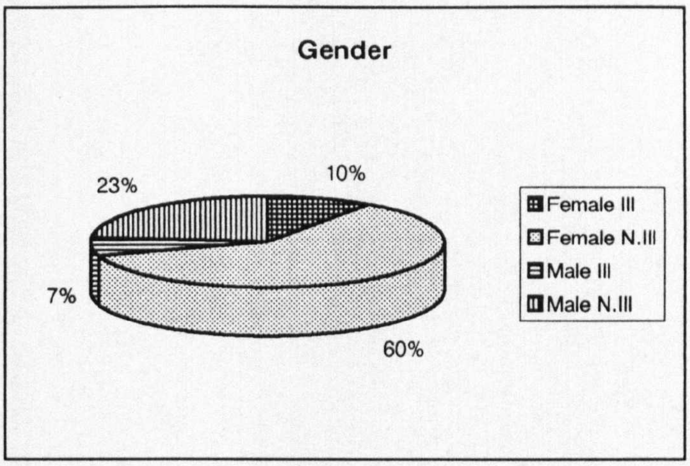

Figure 6.b.8 - Gender vs Illness

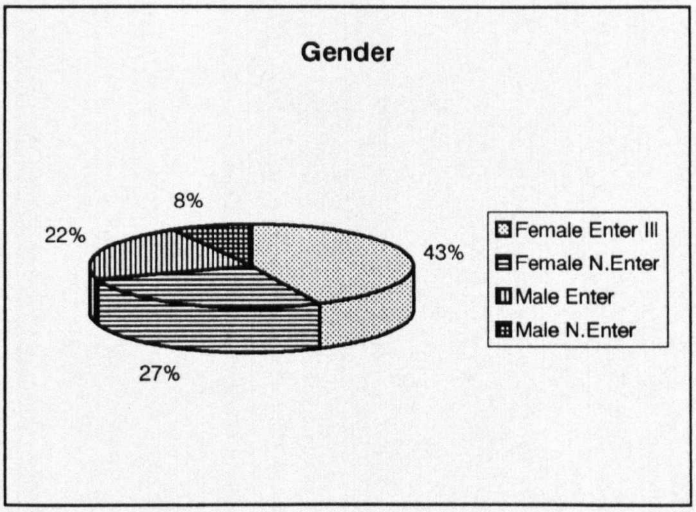

Figure 6.b.9 - Gender vs Entering the Water

6.b.2.4 Visitor Distribution

The telephone survey yielded a total of 402 visitors (travelled over 10 miles) and 133 locals, suggesting the amenities provided at Barry Island are more attractive to visitors than locals (Figure 6.b.10). Illness rate for visitors was 16% and 18% for locals. However, a higher percentage of visitors (67%) entered the water compared to locals (59%) (Figure 6.b.11).

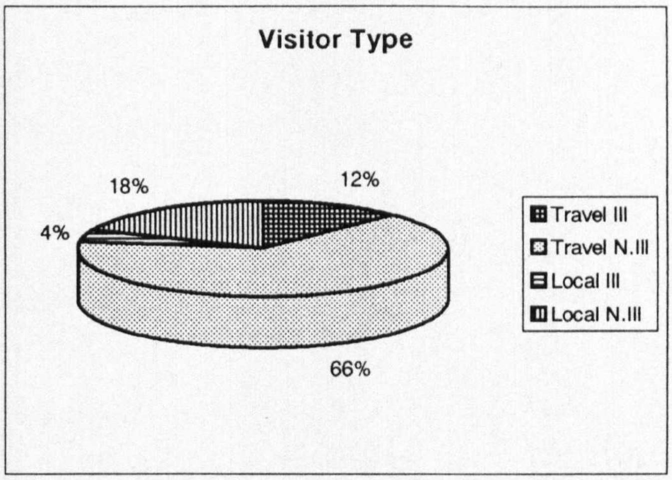

Figure 6.b.10 - Visitor vs Illness

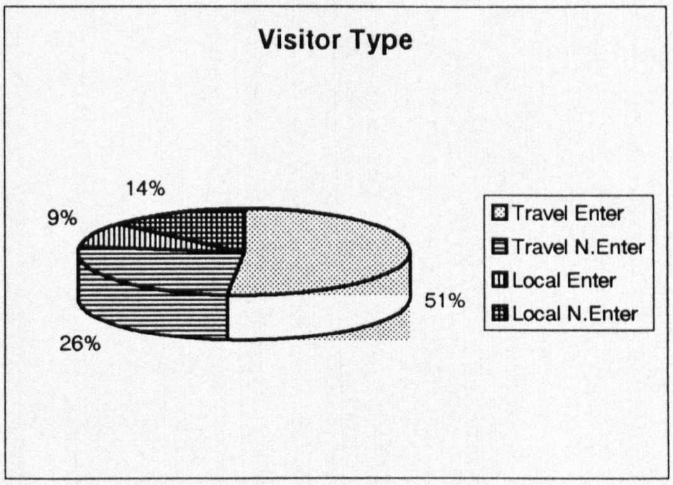

Figure 6.b.11 - Visitor vs Entering the Water

6.b.2.5 Activity and Immersion Distribution

Water activity was broken down into two categories, swim and wade. For the whole sample group there was a greater tendency to wade than swim, although those under 20 years of age were equally split between the two. A simple risk ratio showed swimmers to be 1.8 times more susceptible to illness than waders and those who immersed their head were 1.7 times more at risk of experiencing an illness. Only a quarter (26%) who entered the water immersed their heads.

6.b.2.6 Illness Against Food

Certain food stuffs are considered to present a higher than normal risk to health. A list of high risk foods was drawn up and tabulated using Dillon and Griffiths (1995), Jones *et al.,* (1993), Alexander and Heaven (1990) as sources of information. Insufficient data prevented inclusion of these foods in the logistic model, except for consumption of burgers. Of all the foods burgers are a fast food item readily available at the beach and considered to be a high risk food (Rees *pers.comm.,* 1997). Phillip *et al.,* (1985) also controlled for foods bought onsite in the investigation on risk of illness among snorkellers at Bristol Docks.

Figure 6.b.12 displays a comparison of the distribution of foods eaten by three categories of respondents, showing the percentage of respondents within each series to have eaten the individual foods. The first category (Cases) consisted of individuals who were exposed to the water and were ill. The second category (Controls) consisted of individuals who were not exposed to the water and were not ill. The third category (Exposed and Not Ill) consisted of individuals who were exposed to the water and were not ill. No explicit differences in eating habits is obvious between the three groups, implying food not to be a factor in the higher prevalence of illness in the Cases group.

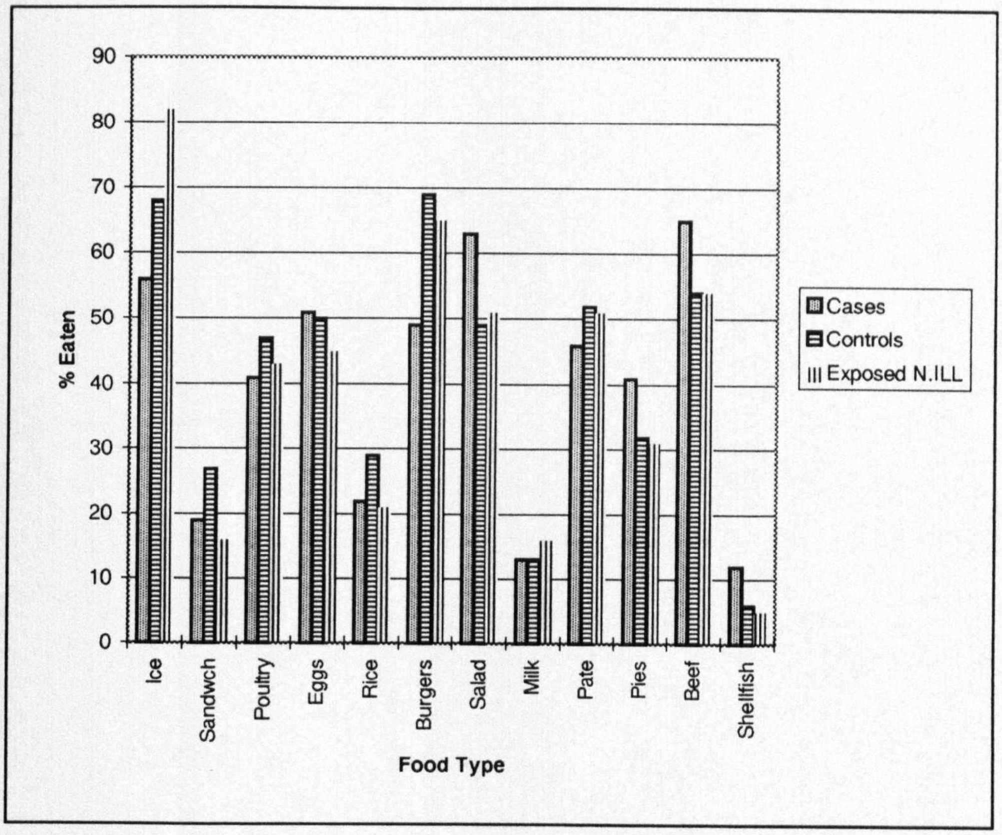

Figure 6.b.12 - Distribution of Foods Eaten

6.b.2.7 Symptom Rates Against Entering the Water

Limited data numbers prevented investigating specific illnesses using the logistic model (see Section 6.b.5). Association between health risk from bathing was centred on whether the illness was present or absent. This approach was recommended by Lightfoot (1989) who suggested considering either one or two types of illness only.

However, to get a feel for prevalence of symptoms among participants in the study, respondents were required to indicate whether they had contracted one or more of a list of symptoms associated with bathing in sewage contaminated seawater. Principal symptoms which show prevalence in exposure groups are discussed in Section 6.3. Symptoms derived here are based on studies by Alexander and Heaven (1990), Balarajan (1992), and Jones *et al.,* (1993). Figure 6.b.13 compares the distributions of three groups:

(1992), and Jones *et al.,* (1993). Figure 6.b.13 compares the distributions of three groups:

i. Cases, which include those participants who were exposed to the water and contracted one of the listed symptoms in the 10 days post beach interview.
ii. Controls, used as a background to compare the Cases against. The Controls were the respondents who had not entered the water either on the day at the beach or within the previous 3 days and had not received an illness following their beach interview. The symptom rates listed were those experienced within 3 days prior to the beach interview.
iii. Exposure, which included those who had entered the water on the day of the beach interview but had not incurred an illness within the following 10 days. Their symptoms were experienced in the 3 days prior to the beach study.

The prevalence of symptoms rates in the Cases group was significantly higher than both the Controls and Exposure groups, except for aching arms, skin rashes and ulcer and headaches. The most obvious elevation in symptom rates for the Cases group against the other two groups were for gastrointestinal related illnesses. The symptoms included stomach pains, nausea, vomiting and diarrhoea (data not available for groups 2 and 3). These findings are in agreement with by Cabelli (*et al.,* 1982; 1983) and Pike (1994) who also found gastrointestinal symptoms to be the most significantly related to illness derived from exposure to contaminated seawater.

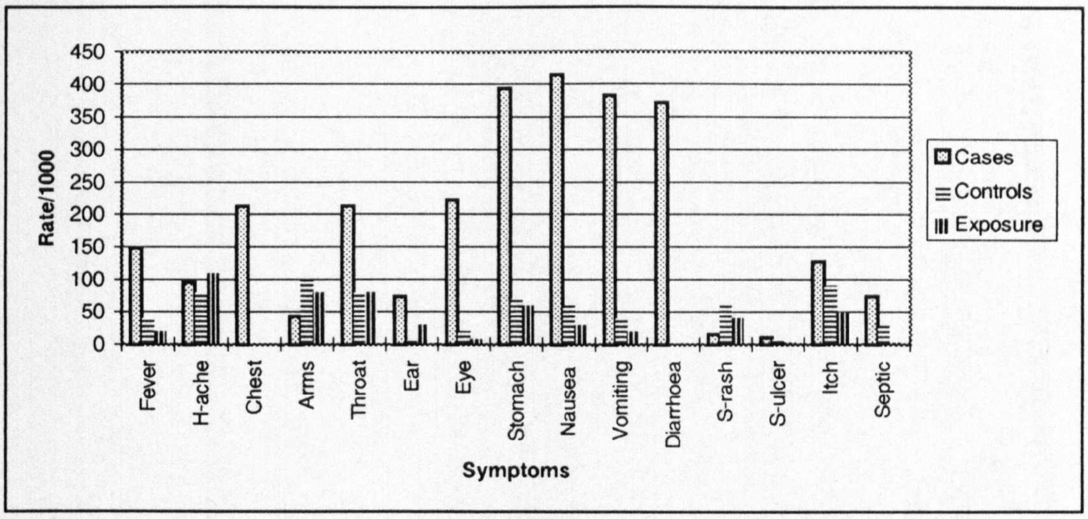

Figure 6.b.13 - Symptoms Rates for Cases vs Controls

6.b.2.7.1 Major Symptoms

To simplify the data, major symptoms were grouped into four categories, by grouping similar symptoms:

1. Fever
2. E.E.T. - ear, eye and throat
3. G.I. - gastrointestinal symptoms including stomach, nausea and vomiting. Diarrhoea was treated separately, explained above. The data describes the Cases.
4. Skin - skin rash, skin ulcer, septic tissue.

The Cases experienced higher rates of illness over both Controls and Exposed groups, except for the Skin category (Figure 6.b.14). Gastrointestinal symptoms were by far the most apparent illness displayed by the Cases, supported by very high rates of diarrhoea.

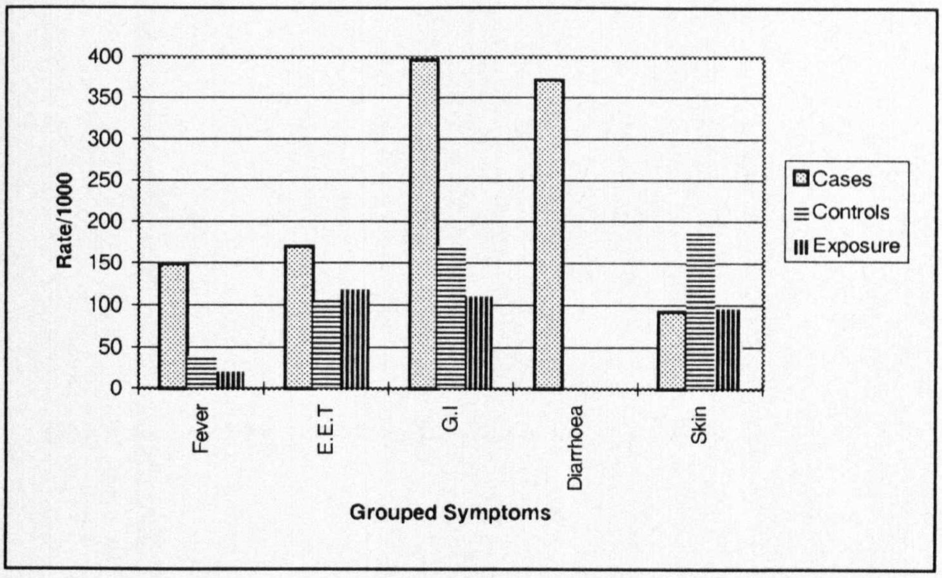

Figure 6.b.14 - Major Symptom Rates Cases vs Control and Groups 1 and 2

6.b.2.7.2 Cases

Gastrointestinal symptoms, explained above, are the most common experienced by bathers exposed to sewage contaminated coastal waters (Cabelli, *et al.,* 1982; Balarajan *et al.,* 1991; Pike, 1994). Jones *et al.* (1993) used gastrointestinal symptoms as his main illness in his logistic modelling. Figures 6.b.13 and 6.b.14 represented self-reported symptoms, i.e. no medical foundation to back them up. Figure 6.b.15 shows the total self-reported gastrointestinal symptoms reported for the Cases group against incidence of gastrointestinal illness which required either a visit to the doctor, hospital or involved medication. Cabelli *et al.* (1982) used this system to verify the seriousness of symptoms experienced, and labelled gastrointestinal symptoms that required medical attention highly credible gastrointestinal symptoms (HCGI). The rates of illness were similar between all gastrointestinal symptoms for both Total and HCGI symptoms. HCGI were approximately one quarter of the Total symptoms reported. However, Total symptoms should not be disregarded, because invariably they are only effective for a short period of time and do not warrant medical attention (HMSO, 1990b).

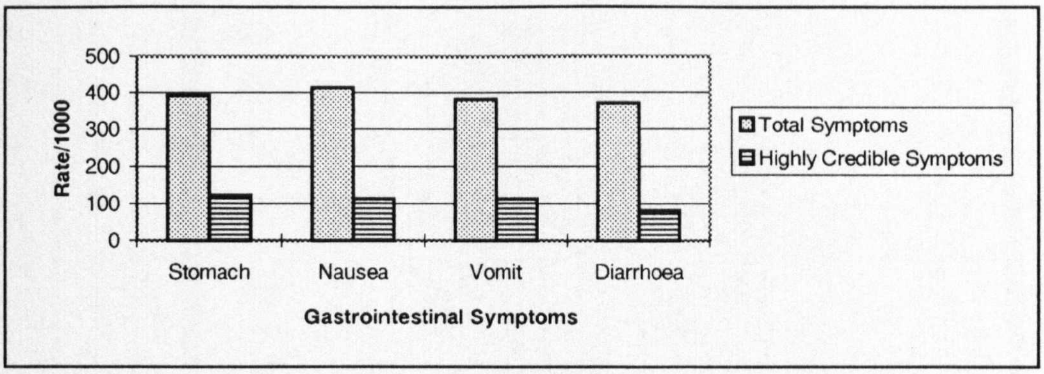

Figure 6.b.15 - Rates of HCGI and Total Gastrointestinal Symptoms for Cases

6.b.3 Health Risk Assessment

The health risk data has been analysed using three statistical techniques described in Chapter 6, Section 6.2. The contingency Table 6.b.1 shows the relative ratio of exposure and illness.

	Ill	Not Ill
Entered water	94	291
Did not enter	3	197

Table 6.b.1 Exposure versus Illness

6.b.4 Chi-Square Analysis (χ^2)

Chi-square analysis χ^2 was used to establish if an association between risk of illness and exposure to seawater existed. Various researchers have used this statistical technique for establishing strength of association between water users and illness rates compared to non-water users, including Phillip *et al.*, (1985), Lightfoot (1989), and Alexander and Heaven (1990).

Jandel Scientific Sigma Stat statistical package (1995) was used to calculate the following χ^2 values. The two characteristics that define the contingency Table 6.b.1 above are significantly related (P = <0.001). Results show that a strong association is apparent between increase in illness rate and exposure to seawater (Nelson and Williams, 1997).

6.b.5 Odds and the Odds Ratio (ψ)

Following from the χ^2 analysis, which proved a significant association between Entering the water (Enter) and Illness, the odds ratio (ψ) gave an evaluation of risk (Nelson *et al.*, in press (b)). A crude estimation of the extent to which exposure had an effect on illness was computed from Table 6.b.1. The odds ratio showed a significant elevation in symptom rates (21.2) among those that entered the water compared to the control group of non-entered (Nelson *et al.*, in press (b)). This result takes no account of risk variables. A relative comparison of illness probabilities were achieved through stratification by age, gender, visitor type and socio-economic status. These values are raw odds ratios adjusted for each variable independently. However, a full analysis controlling for all confounding variables is carried out using multiple logistic regression (Section 6.b.5). Selection of covariates controlled for in the following sets of statistical analysis were derived from previous studies, also listed in Section 6.b.5.

$$\underline{\psi = 21.21 \text{ 95\% C.Interval } (6.65, 67.6)}$$

6.b.5.1 Stratified by Age (ψ)

Results for the odds ratios for each age category are shown below. The questionnaire categorised age into 6 age groups of 10 year ranges. The participants were required to indicate which group their age fell into by selecting the appropriate box. To avoid zero entries, which occurred in some of the unexposed groups, the 6 age group categories were collapsed down, forming 3 sets, 0-19, 20-39 and 40+.

It is apparent that the older age category were least at risk from swimming, followed by the 0-19 age category. The most susceptible age group to illness were the mid-age group 20-39, (Table 6.b.2).

	AGE					
	0=19		20=39		40=60+	
	ILL 1	N.ILL 2	ILL 1	N.ILL 2	ILL 1	N.ILL 2
ENTER 1	48	126	26	85	13	76
N.ENTER 2	1	55	1	89	1	64

Table 6.b.2 Age vs. Illness

ψ_{0-19} = 20.95 95% C.Interval (2.81, 150.33)

ψ_{20-39} = 27.22 95% C.Interval (3.61, 205.06)

ψ_{40-60+} = 10.95 95% C.Interval (1.39, 86.0)

6.b.5.2 Stratified by Gender (ψ)

The odds ratios for males and females were 19.9 and 21.6 indicating that neither gender group have a higher or lesser risk of illness from swimming in the sea, relative to each other (Table 6.b.3).

	GENDER			
	Male		Female	
	ILL 1	N.ILL 2	ILL 1	N.ILL 2
ENTER 1	39	90	55	197
N.ENTER 2	1	46	2	155

Table 6.b.3 Gender vs. Illness

ψ_{male} = *19.9* 95% C.Interval (2.65, 149.49)

ψ_{female} = *21.6* 95% C.Interval (5.19, 89.95)

6.b.5.3 Stratified by Visitor Type (ψ)

In the context of visitor type, the day tripper represented beach users who travelled over 10 miles to reach the beach, as opposed to locals who lived in closer proximity to the beach (Table 6.b.4). No significant difference in odds ratio was apparent between the two groups.

	VISITOR TYPE			
	Day tripper		Local	
	ILL 1	N.ILL 2	ILL 1	N.ILL 2
ENTER 1	70	232	24	55
N.ENTER 2	2	148	1	53

Table 6.b.4 Visitor Type vs. Illness

$\psi_{day\ tripper}$ = *22.33* 95% C.Interval (5.39, 92.44)

ψ_{local} = *23.13* 95% C.Interval (3.02, 177.12)

6.b.5.4 Stratified by Socio-Economic Status (ψ)

Stratification of socio-economic status (SES) resulted in zero entries in the unexposed group which produced infinite odds ratios between exposed and unexposed categories with regard to illness. In contrast to age it was difficult to collapse the categories for SES. To run the logistic model it was necessary to include SES (see Section 6.b.5). The retired group was collapsed with housewives and the unemployed were collapsed with students. Selection of these categories was mostly arbitrary. Although these groups do not have any obvious connection, their odds ratios are calculated below for completeness (Table 6.b.5). The employed category had a very similar odds ratio to the housewife and retired group with values 16.67 and 16.58 respectively. The students and unemployed group had a higher odds ratio of 21.78.

	Employed		H.Wife & Retired		Student & Unemployed	
	ILL 1	N.ILL 2	ILL 1	N.ILL 2	ILL 1	N.ILL 2
ENTER 1	15	81	20	76	59	130
N.ENTER 2	1	90	1	63	1	48

Table 6.b.5 Socio-Economic Status Visitor Type vs. Illness

ψ_{emp} = 16.67 95% C.Interval (2.15, 129.03)

$\psi_{H.wife/ret.}$ = 16.58 95% C.Interval (2.16, 126.0)

$\psi_{Stu./unemp.}$ = 21.78 95% C.Interval (2.94, 161.59);

6.b.6 Mantel Haenszel

The Mantel Haenszel method provides a summary odds ratio ψ_{mh} from a series of 2x2 contingency tables obtained from the stratification of the Cases and Controls on the basis of one or more variables, e.g. a stratification of Cases and Controls by the variable age.

Further the Mantel Haenszel technique permits the assessment of individual and joint effects of a set of risk factors, with adjustment for confounding by one or more variables (Hosmer and Lemeshow, 1989). In general, the computed odds ratios estimates for each stratum will be different due to sampling, confounding effects or even interaction effects. However, correct estimates of the effect of the risk factors by the Mantel Haenszel technique will only be obtained when the odds ratio is constant across the stratum, i.e. in the absence of interaction effects. The criteria used to decide whether a variable has a confounding effect on a risk variable is to compare the value of ψ_{mh} with the raw odds ratio for the risk variables enter versus the outcome variable illness. Section 6.b.3 outlines the inherent problems with using this technique if zero entries are observed in the contingency tables. Shlesselman (1992) gives approximate methods by using a ½ correction factor to overcome these zero entries which lead to the computation of infinite values.

6.b.6.1 Stratified by Age (ψ_{mh})

In the case of the Age covariate two calculations of ψ_{mh} were performed. The first attempt retaining the full 6 categories of age (0-9; 10-19; 20-29; 30-39; 40-49 and >50) required use of the ½ correction factor suggested by Shlesselman (1992). A second approach to overcome zero entries when computing ψ_{mh} was used which collapsed the six categories of age down to three categories (0-19; 20-39; 40-60+). The two resultant ψ_{mh} were very close, although their 95% confidence intervals were dissimilar. The difference between the two sets of confidence intervals is due to variance between their respective standard errors. It is possible that the correction factor dealing with zero entries is not as reliable as computation of ψ_{mh} when data has no zero entries.

The odds ratios for the data stratified according to Age 0-19; 20-39 and 40-60+ were 20.95, 27.22 and 10.95 respectively. The corresponding value of ψ_{mh} was 16.07 with 95% Confidence Interval (2.7, 95.51). The raw odds ratio ψ for the risk variable Enter versus Illness was 21.21 with 95% Confidence Interval (6.65, 67.6). The difference between ψ and ψ_{mh} were considered significant and indicated age to be a confounding factor.

6.b.6.2 Stratified by Gender (ψ_{mh})

The odds ratios for the data stratified according to Gender were 19.9 for males and 21.6 for females. The corresponding value of ψ_{mh} was 21.7 with 95% Confidence Interval (4.0, 117.58). This value was not considered significantly different from the raw odds ratio ψ for the risk variable Enter versus Illness, 21.21 with 95% Confidence Interval (6.65, 67.6), which indicated Gender not to be a confounding factor.

6.b.6.3 Stratified by Visitor (ψ_{mh})

The odds ratios for the data stratified according to Visitor type were 22.33 for travellers and 23.13 for locals. The corresponding value of ψ_{mh} was 18.38 with 95% Confidence Interval (3.78, 89.32). The raw odds ratio ψ for the risk variable Enter versus Illness was 21.21 with 95% Confidence Interval (6.65, 67.6). The difference between ψ and ψ_{mh} were considered significant and indicated Visitor type to be a confounding factor.

6.b.6.4 Stratified by Socio-Economic Status (ψ_{mh})

The odds ratios for the data stratified according to Socio-economic Status: employed, house-wife/retired and student/unemployed were 16.67, 16.58 and 21.78 respectively. The corresponding value of ψ_{mh} was 18.56 with 95% Confidence Interval (14.16, 24.32). The raw odds ratio ψ for the risk variable Enter versus Illness was 21.21 with 95% Confidence Interval (6.65, 67.6). The difference between ψ and ψ_{mh} were considered significant and indicated Socio-economic Status to be a confounding factor.

6.b.6.5 Data Limitations

The Mantel Haenszel Method uses a weighted summary estimate to adjust for confounding factors, as can be observed above. However, to control for all of the selected potential confounding variables would mean considerable calculations across many levels of stratification. Multiple logistic regression overcomes this problem and also accounts for interaction effects, which the Mantel Haenszel Method does not. Statistica for Windows Release 4.1(Statsoft, 1993) statistical package was used for the computation.

6.b.6.6 Summary of Mantel Haenszel Odds Ratio

Data used to calculate the Mantel Haenszel summary odds ratios were derived from Tables 6.b.2, 6.b.3, 6.b.4 and 6.b.5. Table 6.b.6 summarises the Mantel Haenszel odds ratios for the confounding factors Age, Gender, Visitor type and Socio-economic Status.

Covariate	**Mantel Haenszel ψ_{mh}**	**95% Confidence Interval**
Age (6 categories)	15.94	5.59, 45.57
Age (3 categories)	16.07	2.7, 95.51
Gender	21.70	4.0, 117.58
Visitor type	18.38	3.78, 89.32
Socio-economic status	18.56	14.16, 24.32

Table 6.b.6 Mantel Haenszel Odds Ratios

6.b.7 Multiple Logisitic Regression

Previous work on marine health risk analysis has been conducted using various statistical techniques (Phillip *et al.,* 1985; Alexander and Heaven, 1990; Kay *et al.,* 1994). A notable turning point in the derivation of water quality standards based on statistical evidence of health risk from bathing came from Cabelli's *et al.* (1982) work in the 1970s. His results formed the foundation for standards set by the US Environmental Protection Agency (EPA) (see Section 3.7.3). Least squares linear regression was used to examine the relationship between illness and indicator density by pooling the data, clustering points around set concentrations as opposed to using trial days as a base for measurement. Fleisher (1992) argued that Cabelli's analysis failed to account for variation between sites which he proved to have a significant effect on illness rates. In addition the work did not consider confounding factors or interaction effects, reported by Lightfoot (1989). Fleisher (1992) re-examined the data obtained by Cabelli and co-workers applying logistic regression, which accounts for confounding factors (WHO/UNEP, 1991).

Multiple logistic regression (MLR) described in the Chapter 5, Section 5.2.4 is the most powerful statistical technique used in this study to investigate the health effects from bathing in sewage contaminated water. The main reason for selecting the technique is that it gives accurate estimates of odds ratios for a selected risk variable against illness, whilst controlling for other confounding and interaction effects. Breslow and Day (1980) recommend using Multiple Logistic Regression analysis in epidemiological studies seeking to quantify exposure-disease associations. Most recent studies have opted to utilise this technique (Lightfoot, 1989; Fleisher, 1992; Kay *et al.,* 1994; WRc, 1996b), ideally suited for dealing with a binary response variable. Logistic regression which overcomes the problems encountered with the Mantel Haenszel Method has also been endorsed by the WHO/UNEP (1993) for prospective microbiological-epidemiological studies related to water quality.

Five logistic models were generated based on data obtained from the survey carried out in 1995. The first model institutes the variable Enter as the risk factor. Enter is a dichotomous variable differentiating between those that were exposed to the water compared to a non-exposed control group. Models 2 and 3 were generated for only those that entered the water to investigate if the risk of illness depended on the level of faecal indicator density. Model 2 looked at *E.coli* and model 3 looked at faecal streptococci. Daily geometric mean values were computed for both variables *E.coli* and faecal streptococci over the six day sampling period, which were coded into 5 variables. Age was coded using two variables to define the three collapsed age ranges, 0-19 years, 20-39 years and 40-60+ years and socio-economic status was also coded using two variables to define the three collapsed categories, employed, housewife and retired and students and unemployed. Chapter 5, Section 5.2.4.3 gives a full explanation of the coding procedure

Generation of models 4 and 5 was an attempt to create simplified models using the risk variables *E.coli* (model 4) and faecal streptococci (model 5), both independently based on a significant biological cut-off point. Selection of the cut-off point was derived from the EC Bathing Water Directive (CEC, 1976a) using the Mandatory level of 2000/100ml for *E.coli* and Guideline level of 400/100ml for faecal streptococci. At present there is was not a Mandatory standard for faecal streptococci in the current EC Bathing Water Directive (CEC, 1976a). However, the proposed new Directive contains a Mandatory standard of 100 per 100 ml for faecal streptococci (CEC, 1997).

6.b.7.1 Model Building and Comparison with the Raw Odds Ratios

Stage 1 of the model building process added potential confounding variables into the model individually. Depending on their effect on the deviance, they were either selected or dropped from inclusion into stage 2 of the model development. The Methods Chapter (Section 5.2.4) explains the relevance of the model deviance and selection criterion reliant upon the resultant probability levels (Section 5.2.5). Stage 2 used the significant confounding variables to build the model by adding them in, in order of highest partial deviance. Choice of appropriate potential confounding variables tested were derived from analysis of prominent epidemiological studies (Phillip *et al.,* 1985; Lightfoot, 1989;

Alexander and Heaven, 1990; Pike, 1994; WRc, 1996b). Data on the participants in the study were obtained via a semi-structured questionnaire (Section 5.4.2).

The main difference in factors chosen for inclusion between model 1 and models 2-5 were the variables Activity which discriminated between waders and swimmers, and Immersion which identified between the swimmers who immersed their head compared to those that did not. Obviously these two variables are not appropriate for model 1. As discussed above limited data meant that the dependent variable illness was based on either presence or absence and not on prevalence of certain symptoms. Also many foods were omitted due to lack of data except for the inclusion of the food Burgers. Fast foods are thought to be risk foods and have been used in previous studies (Phillip *et al.,* 1985). Rees (*pers.comm.,* 1997) suggested selection of the variable *Burger* as a test food. This obviously is not an ideal situation and it would be prudent to further develop the study analysing a wider spectrum of foods, funding permitting. However, these models consider a wider range of confounding variables than other prominent research studies such as Balarajan *et al.,* (1991ref 162) who stated they only adjusted for age and gender in their analysis.

Interaction effects between the main confounding variables *Enter, SES, E3Day, Immerse, Age1&Age2* and *Gender* were investigated. In the case of all 5 models no interaction effects were observed. Statistica for Windows (Statsoft, 1993) statistical software was utilised in calculating the logistic models.

6.b.7.2 Development of Model 1

Development of logistic model 1 was based around *Ill* being the outcome variable and *Enter* the main risk variable. The odds of illness from entering the water were 21.94, 95% confidence interval of 6.85, 70.29, not controlling for any confounding factors. This result was in line with the odds calculated from the contingency tables 21.21, 95% confidence interval of 6.65, 67.6 (Table 6.b.1). The variable *E3Day* discriminated between those that had entered the water in the three days preceding the beach interview and those who had not. This variable displayed the highest partial deviance. Socio-economic status (*SES*) proved to give the second largest partial deviance followed by the variable Immerse which related to those who had dipped their head in the water while swimming and Age (*Age1&Age2*), which was the fourth variable to show statistical significance for inclusion in the model. Age was split into 3 collapsed age ranges outlined above, represented by the two dummy variables Age1 and Age2. The probability derived from the partial deviance for gender (*G*) was 0.09 meaning it just failed significance at the p=0.05 level, but was included in the model for reasons outlined below. Visitor (*Visitor)* type and whether or not the respondent had eaten a burger (*Burger*) on their day at the beach were not significant factors, although the odds ratio from the contingency analysis showed visitor status to be important in determining health risk.

The model *Enter*, *SES* and *E3Day* was significant (p = 0.05). However, although independently *Age* was significant at the p = 0.05 level, addition into the model only gave a significance level of p = 0.18. It was felt that even though it is impossible to justify maintenance of *Age* in the model on statistical grounds it should be included on the basis of further development of the model for the investigation of interaction effects, future prediction purposes and possible application of other modelling techniques. For similar reasons gender was also maintained in the model. Hosmer and Lemeshow (1989) stated that rigidly adhering to a significance level of p = 0.05 is not advocated, and that 'expert' selection of variables can take precedence over statistical significance. The intuitive inclusion of *Age* and *Gender* can be justified on biological grounds, for future prediction puruposes or use in further statistical modelling; *Age* also showed to be an important factor in the stratified odds ratios. In addition leaving variables *Age* and *Gender* in the model had little effect on the resultant odds ratios, changing from 32.54 with 95% confidence intervals of (9.33, 113.43) to 31.37 with 95% confidence intervals of (9.01, 109.26), a difference of 1.17. It was not the objective of this research to use logistic regression for prediction purposes, only to calculate the odds ratios. However, other authors have used logistic regression for prediction, for example Jones (*et al.,* 1993). They came up with a model used in predicting the probability of objective gastrointestinal symptoms against levels of faecal streptococci.

The final model chosen indicated that the odds of contracting an illness from exposure to seawater at Whitmore to be 31.37 times more likely than the non-exposed control group. This is in contrast to the original raw odds ratio of *Enter vs Illness* which found the odds ratio to be 21.21. Therefore, it is obvious that the confounding variables had a significant attenuating effect on the risk variable *Enter*, increasing by approximately 50% when statistically adjusting for their presence. It can be seen that multiple logistic regression is a powerful technique for calculating a true odds ratio adjusted for all confounding variables. The final model for the *Enter vs Illness* is show in Table 6.b.7:

6.b.7.2.1 Model 1

Dependent variable	*Independent variables*	*Odds Ratio*	*95% C.Interval*
Illness	Enter + SES4 + E3Day + Age1&Age2 + G	31.37	(9.0, 109.3)

Table 6.b.7 Model 1 for Enter vs Illness

Table 6.b.8 below shows the development of model 1 by selection and inclusion of covariates based on their partial deviance.

Variable	Dev.	P. Dev.	df	p	Odds	95% CI
Constant	525.54					
Enter	457.00	68.54	1	<0.0001	21.94	6.85, 70.29
Enter + Age1&Age2	448.63	8.37	2	0.0152	19.65	6.12, 63.26
Enter + SES1&SES2	447.1	9.9	2	0.0071	18.28	5.7, 58.89
Enter + E3Day	441.16	15.84	1	0.0001	36.94	11.15, 12.34
Enter + Gen	453.66	3.34	1	0.0676	21.06	6.57, 67.52
Enter + Visitor	455.25	1.75	1	0.1859	22.59	7.04, 72.44
Enter + Burger	454.13	1.88	1	0.1705	21.23	6.63, 68.05
Enter + SES1&SES2 + E3Day	429.82	11.34	2	0.0035	31.22	9.36, 104.13
Enter + SES1&SES2 + E3Day + Age1&Age2	427.09	2.73	2	0.2554	32.10	9.60, 107.39
Enter + SES1&SES2 + E3Day + Age1&Age2 + G	423.61	3.48	1	0.0621	31.38	9.37, 105.03

Table 6.b.8 - Development of Model 1 Showing Odds Ratios for the Enter Variable

6.b.7.2.2 Odds Ratios of Confounding Variables Included in Model 1

Individual odds ratios statistically adjusted for all other covariates can be obtained by taking the exponential of the regression coefficients for each variable separately. Multiple logistic regression was used to generate odds ratios controlling for age, gender, visitor type and SES individually to compare with the odds ratios obtained using the contingency table analysis (Tables 6.b.2, 6.b.3, 6.b.4 and 6.b.5). Results obtained were almost identical, including similar 95% confidence intervals. This substantiates the validity of both techniques as they produce concurrent results. In addition the deviances obtained through running the logistic regression analysis adds weight to the argument that confounding variables are effecting the models. In general, the variables investigated in this research programme are included in interview schedules for most epidemiological/microbiological studies. These studies in the main address the effect of age, presence of gastrointestinal symptoms, water activity, head immersion, time of immersion and dose response relationships against *E.coli* and faecal streptococci (Cabelli, 1983, *et al.,* 1982; Phillip, 1985; ; Balarajan, 1992; Alexander and Heaven, 1991). However, Jones *et al.,* (1993) found non-bathing water-related risk factors not to confound the risk variable faecal streptococci.

6.b.7.3 Development of Models 2 and 3

As stated, the aim of developing logistic models defining faecal bacteria as the risk variables was to investigate whether a dose response relationship existed between indicator concentration and illness rate. The risk variables *E.coli* (EX1-EX5) and faecal streptococci (SX1-SX5) are represented by Models 2 and 3. Running *E.coli* vs. illness and faecal streptococci vs. illness showed that in both cases the partial deviances were not significant at the p = 0.05 level. The models were further developed to investigate whether the inclusion of other confounding variables affected the odds ratios. The variables *SES (SES1&SES2), E3Day, Immerse* and *Age (Age1&Age2)* were added to models 2 and 3, partial deviances proving significant at the p=0.05 level. The partial deviance for the variable *Gender* produced a p value of 0.087. Again *Gender* was included in both models 2 and 3 on biological grounds, similar to model 1 on the assumption that it may prove useful for future prediction purposes. The addition of *Gender* had little effect on the resultant odds ratios. The final odds ratios for EX1-EX5 and SX1-SX5 are discussed below, displayed in Tables 6.b.9 and 6.b.10. Tables 6.b.3 and 6.b.4 show the final selection of covariates included in Models 2 and 3.

6.b.7.3.1 Model 2

Dependent variable	*Independent variables*
Illness	EX + SES1&SES2 + E3Day + Immerse + Age1&Age2 + G

Table 6.b.9 - Model 2 *E.coli* vs. Illness

6.b.7.3.2 Model 3

Dependent variable	*Independent variables*
Illness	SX + SES1&SES2 + E3Day + Immerse + Age1&Age2 + G

Table 6.b.10 - Model 3 faecal streptococci vs. Illness

Table 6.b.11 and 6.b.12 shows the development of models 2 and 3 by selection and inclusion of covariates based on their partial deviance.

Variable	Dev.	P. Dev.	Df	p
E.Coli coded Ex1-Ex5 (EX)				
Constant	425.73			
EX	420.62	5.11	5	0.4025
EX + Age1,Age2 (Age)	413.40	7.22	2	0.0271
EX + SES1&SES2	411.69	8.93	2	0.0115
EX+E3Day	408.09	12.53	1	0.0004
EX+Gender (G)	417.79	2.83	1	0.0925
EX + Visitor	419.18	1.44	1	0.2301
EX + Burger	418.05	2.57	1	0. 1087
EX+Immerse (Imrs.)	412.86	7.76	1	0.0053
EX + Activity	419.96	0.66	1	0.41626
EX + E3Day + SES1&SES2	397.77	10.32	2	0.0058
EX + E3Day + SES1&SES2 + Immerse	389.49	8.29	1	0.004
EX + E3Day + SES1&SES2 + Immerse + Age1&Age2	387.05	2.44	2	0.1183
Model 2				
EX + E3Day + SES1&SES2 + Immerse + Age1&Age2 + G	385.30	1.75	1	0.1858

Table 6.b.11 Development of Model 2

Variable	Dev.	P. Dev.	df	p
Faecal Streptococci ,Coli coded Sx1-Sx5 (EX)				
Constant	425.73			
SX	420.21	5.52	5	0.3553
SX + Age1,Age2 (Age)	412.94	7.26	2	0.0265
SX + SES1&SES2	410.86	9.35	2	0.0094
SX+E3Day	407.87	12.34	1	0.0004
SX+Gender (G)	417.28	2.92	1	0.0893
SX + Visitor	418.81	1.40	1	0.2371
SX + Burger	417.67	2.54	1	0.1112
SX+Immerse (Imrs.)	412.45	7.76	1	0.0053
SX + Activity	419.54	0.67	1	0.4126
SX + E3Day + SES1&SES2	397.51	10.36	1	0.0056
SX + E3Day + SES1&SES2 + Immerse	389.23	8.2	1	0.004
SX + E3Day + SES1&SES2 + Immerse + Age1&Age2	386.80	2.42	2	0.2978
Model 2				
EX + E3Day + SES1&SES2 + Immerse + Age1&Age2 + G	385.01	1.79	1	0.181

Table 6.b.12 Development of Model 3

6.b.7.3.3 Odds Ratios of Confounding Variables Included in Models 2 and 3

The variables EX1-EX5 and SX1-SX5 are all relative to a selected reference value. The reference values chosen were the lowest indicator levels for *E.coli* (1052) and faecal streptococci (260). If a relationship between indicator density and illness rate was present then a numeric progression in the odds ratios would be observed between EX1-EX5 and SX1-SX5. However, this is not the case, none of the odds ratios show an increased risk of illness for the varying levels of *E.coli* and faecal streptococci compared to the reference levels (i.e. *E.coli* level 1052 per 100ml and faecal streptococci level 260 per 100ml) after controlling for confounding (Tables 6.b.13 and 6.b.14). Also the confidence limits calculated for the odds ratios EX1-EX5 and SX1-SX5 (Tables 6.b.13 and 6.b.14) all fall below one, supporting the finding that there is no increased risk of illness compared to the corresponding reference levels of *E.coli* and faecal streptococci. Therefore, it can be concluded that these results do not suggest a dose response relationship between bacterial concentration and reporting of illness rates.

Variable	Odds Ratio	95% Confidence Limits
EX1	1.16	0.56, 2.41
EX2	0.86	0.39, 1.9
EX3	1.02	0.39, 2.68
EX4	0.78	0.33, 1.84
EX5	1.33	0.58, 3.05

Table 6.b.13 Odds Ratios Model 2

Variable	Odds Ratio	95% Confidence Limits
SX1	0.88	0.33, 2.36
SX2	0.86	0.41, 1.79
SX3	0.74	0.33, 1.68
SX4	1.19	0.51, 2.79
SX5	0.65	0.28, 1.56

Table 6.b.14 Odds Ratios Model 3

6.b.7.4 Development of Models 4 and 5

Tables 6.b.17 and 6.b.18 show the development of model 4 (Table 6.b.15) and model 5 (Table 6.b.16) which consider *E.coli* and faecal streptococci (F.S.) respectively as the risk variables, and investigate whether confounding variables have any effect on the resultant odds ratios. Both models show the odds ratios to be < 1. Therefore, this suggests that there is no greater risk associated with increasing levels of *E.coli* and faecal streptococci with respect to illness rates from exposure to seawater. These results support the findings of models 2 and 3.

6.b.7.4.1 Model 4

Dependent variable	*Independent variables*	ψ *E.coli*	*95% C.Interval*
Illness	*E.coli* + E3Day + Immerse + SES1&SES2 + Age1Age2 + Gen	0.576	0.31, 1.08

Table 6.b.15 - Model 4 for *E.coli* vs Illness

6.b.7.4.2 Model 5

Dependent variable	*Independent variables*	ψ F.S.	*95% C.Interval*
Illness	F.S. + E3Day + Immerse + SES1&SES2 + Age1Age2 + Gen	0.76	0.46, 1.25

Table 6.b.16 - Model 5 for Faecal Streptococci vs Illness

Variable	Dev.	P. Dev.	Df	p	Odds	95% CI
Constant	425.73					
E.coli	418.21	7.515	1	0.0061	0.51	0.32, 0.83
E.coli + Age1&Age2	410.47	15.26	2	0.0005	0.53	0.33, 0.86
E.coli + SES1&SES2	409.80	15.94	2	0.0003	0.54	0.33, 0.88
E.coli + E3Day	403.39	22.34	1	<0.0001	0.50	0.30, 0.81
E.coli + Gen	415.88	9.85	1	0.017	0.53	0.33, 0.86
E.coli + Visitor	415.88	9.85	1	0.017	0.50	0.31, 0.80
E.coli + Burger	415.73	10.0	1	0.016	0.51	0.32, 0.83
E.coli + Immerse	412.16	13.57	1	0.0002	0.54	0.33, 0.87
E.coli + Activity	417.74	7.99	1	0.047	0.52	0.32, 0.84
E.coli + E3Day + Immerse	394.59	8.79	1	0.003	0.52	0.32, 0.85
E.coli + E3Day + Immerse + SES1&SES2	386.67	7.92	2	0.0191	0.55	0.34, 0.91
E.coli + E3Day + Immerse + SES1&SES2 + Age1&Age2	383.98	2.69	2	0.2606	0.56	0.34, 0.92
E.coli + E3Day + Immerse + SES1&SES2 + Age1&Age2 + Burger	383.18	0.80	1	0.3707	0.55	0.33, 0.91
E.coli + E3Day + Immerse + SES1&SES2 + Age1&Age2 + Gen	382.51	1.47	1	0.0621	0.58	0.31, 1.08,
E.coli + E3Day + Immerse + SES1&SES2 + Age1&Age2 + Visitor	381.27	1.91	1	0.2252	0.56	0.33, 0.93
E.coli + E3Day + Immerse + SES1&SES2 + Age1&Age2 + Activity	381.6487	0.8581	1	0.3543	0.59	0.35, 0.98

Table 6.b.17 - Development of Model 4 Showing Odds Ratios for the *E.coli* Variable

Variable	Dev.	P. Dev.	Df	p	Odds	95% CI
Constant	425.73					
F.S.	423.06	2.66	1	0.1092	0.51	0.32, 0.83
F.S. + Age1&Age2	414.94	8.12	2	0.0173	0.70	0.44, 1.12
F.S. + SES1&SES2	414.10	8.96	2	0.0113	0.72	0.45, 1.15
F.S. + E3Day	408.51	14.56	1	0.0001	0.67	0.41, 1.08
F.S. + Gen	420.27	2.79	1	0.0949	0.70	0.43, 1.12
F.S. + Visitor	421.78	1.29	1	0.257	0.70	0.43, 1.11
F.S. + Burger	420.45	2.62	1	0.1057	0.67	0.42, 1.07
F.S. + Immerse	416.56	6.50	1	0.0108	0.71	0.44, 1.14
F.S. + Activity	422.54	0.52	1	0.4693	0.70	0.43, 1.11
F.S. + E3Day + Immerse	399.20	9.31	1	0.023	0.70	0.43, 1.34
F.S. + E3Day + Immerse + SES1&SES2	390.64	8.56	2	0.0139	0.74	0.45, 1.21
F.S. + E3Day + Immerse + SES1&SES2 + Age1&Age2	387.70	2.94	2	0.23	0.75	0.45, 1.23
F.S. + E3Day + Immerse + SES1&SES2 + Age1&Age2 + Gen	385.88	1.82	1	0.1772	0.76	0.46, 1.25

Table 6.b.18 - Development of Model 5 Showing Odds Ratios for the Faecal Streptococci Variable

6.b.7.5 Investigation of a Linear Relationship Between Faecal Indicators and Illness

The three logistic models investigating a dose response relationship between increasing levels of faecal indicator showed no increased risk. As discussed in Section 6.b.5 linear logistic regression has been one of the preferred statistical modelling techniques dealing with a dichotomous output variable in epidemiological/microbiological studies (Lighfoot, 1989; Fleisher, 1992 ; Jones *et al.*, 1993; WRc, 1996b). Earlier studies used linear techniques to analyse their data, for example Cabelli *et al.*, (1982). Although these techniques are not as robust as logistic regression, not accounting for confounding factors, they do provide a visual description of the data. Figures 6.b.16 and 6.b.17 provide geometric mean plots of *E.coli* and faecal streptococci densities against rates of illness per 1000 persons (Nelson and Williams, 1997). The six levels of bacteria shown on both graphs represent the average geometric mean counts of each of the six survey days during 1995. It is apparent that as the level of both *E.coli* and faecal streptococci increase there is a corresponding decrease in illness rates. This lack of positive correlation agrees with the findings from Models 2,3,4 and 5.

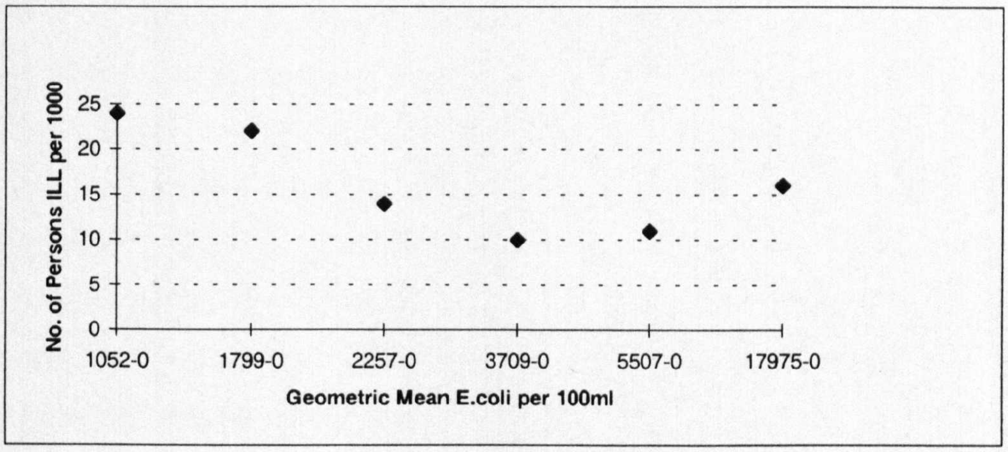

Figure 6.b.16 - Average Geometric Mean Counts of *E.coli*/Survey Day vs. Illness Rates

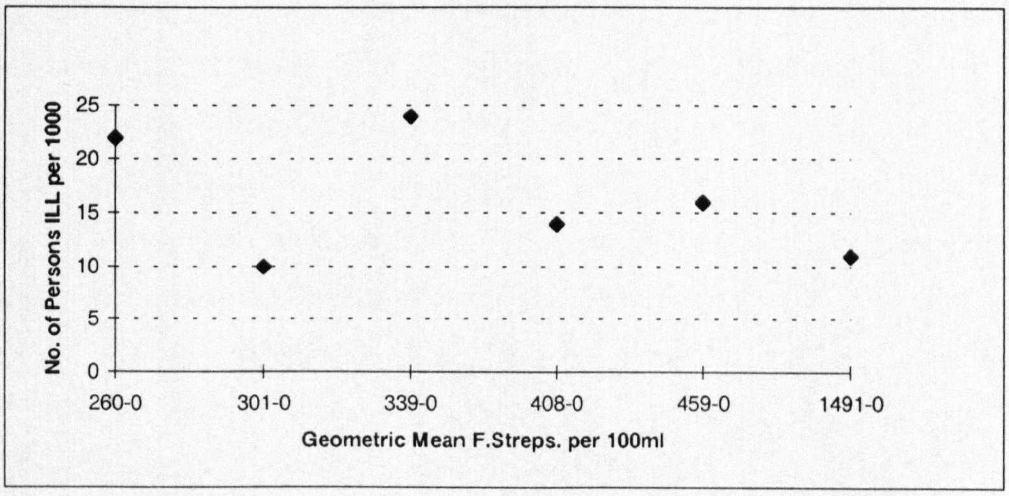

Figure 6.b.17-Average Geometric Mean Counts of Faecal Streptococci/Survey Day vs. Illness Rates

6.b.7.6 Summary of Health Risk Results

Statistical modelling, using the multiple logistic regression technique, showed swimmers to significantly increase their chance of contracting an illness in comparison to non-swimmers. These findings are in agreement with other major studies in the field (Cabelli *et al.,*1982; Phillip *et al.,* 1985; Kay *et al.,* 1994). Multiple logistic regression proved socio-economic status, age and entry into seawater three days prior to the interview day to have confounding effects. Introduction of the variable gender to the logistic model on biological grounds, had little effect on the regression coefficients. It was added to the model to accommodate future prediction modelling and further investigation of other statistical models. No interaction effects were evident similar to work done by Lightfoot (1989) and Kay *et al.* (1994). Logistic regression modelling failed to show any positive correlation between bacterial indicator density and morbidity rates among swimmers, implying a dose response relationship does not exist.

6.b.8 Discussion of Health Risk Analysis

Strong evidence links an increase in reported illness with exposure to faecally contaminated water (Cabelli, 1983; Phillip *et al.,* 1985; Alexander and Heaven, 1991; Pike, 1994; WRc, 1996a) in concurrence with the findings from this study. The final report of the DoE Health Effects study (WRc, 1996a) also found subjects who had had contact with water to be at a higher risk of infection than non-contact subjects. However, no consistency is apparent through the literature identifying an appropriate indicator to model health risk from bathing. For a full discussion on indicators refer Section 3.5. Kay (1994) and Cabelli *et al.* (1982) argued that *E.coli* is an inappropriate indicator of sewage contaminated waters, failing to provide an indication of health risk from bathing. This would be consonant with the lack of association between *E.coli* levels and reported illness in this study. Certain studies have reported to find strong evidence linking faecal streptococci and disease (Cabelli *et al.,* 1982; Fleisher, 1992; Jones *et al.,* 1993). The only study of a similar size which used multiple logistic regression was carried out by Jones *et al.,* (1993), based on a sample of 350. Their study differed by using a controlled cohort study in contrast to the prospective approach employed here. Results of their research produced a predictive mathematical equation to model faecal streptococci density (as a continuous variable) and objective gastrointestinal symptoms against illness. They predicted an increase in illness at around 32 counts of faecal streptococci per 100ml.

Results of this study showed no apparent relationship linking faecal streptococci to increased incidence of illness, in agreement with Lightfoot (1989) who also found no evidence to reveal a relationship between disease and bacterial count. In addition the final WRc report (1996b) which re-analysed data from the DoE Beach Survey work (Pike, 1994) revealed no significant positive relationships between rates of illness and concentrations of bacterial indicators. Further, the report (WRc, 1996b) questioned the validity of Jones *et al.,* (1993) predictive model, claiming the threshold limit of 32 faecal streptococci per 100 ml to be unconvincing. If the work done by Jones *et al.,* (1993) was valid, the concentration of faecal streptococci detected at Barry Island during the 1995 bathing season well exceeds these 32 per 100 ml (maximum geometric mean levels >1400 per 100ml). Therefore it is not possible to directly extrapolate the data back to compare with the results of Jones and his colleagues. It might be that their work is only applicable to low levels of faecal streptococci concentrations. Results obtained for different bathing waters have also been argued to be site specific (HMSO, 1990b; WRc, 1996a), which would invalidate comparison of models. The intention of this research was to purely establish the odds ratio indicating risk and not to produce a predictive model.

There is not currently a standardised protocol for frequency of water sampling. The sampling frequency used in this study exceeded that followed by Lightfoot (1989) and Alexander and Heaven (1990). However, it is possible that the lack of association between faecal indicator density and illness could be that higher levels were in existence but undetected at the sites monitored. Kay *et al.* (1994) claimed that this is a methodological flaw, and to overcome it the microbiological quality of the water should be assigned to each bather at the time and place of bathing. The finances of this research would not permit such an approach. As an alternative Cabelli's (1975) notion was adhered to that a daily geometric mean would be representative of the water quality based on the premise that swimmers tend to swim at various times during the day.

The lack of association between microbiological quality of the water and illness rates amongst swimmers, although site specific, adds to the continuing debate over appropriate indicators of health (Lightfoot, 1989; Pike, 1994). Cartwright (1993) stated that to set achievable standards to protect swimmers in recreational waters, more information was required to understand the relationship between the parameters and disease before expensive measures are spent to improve the quality of bathing waters. Appropriate control measures can only be undertaken when the pathogenesis of disease is better understood. The WHO (1994a) have also stated that more work is needed with well-designed epidemiological studies for the assessment of health risks, both infectious and man-made, that are associated with exposure to different environment hazards. An alternative theory on protection of health from bathing in recreational waters, which would appear more robust, is destruction of pathogens at source (Rees, *pers.comm.*, 1995). A cost-benefit analysis would have to be conducted to justify this method.

The modelling exercise used in this study utilises linear logistic regression. For a fuller analysis other models might be investigated, based on for example an exponential function, suggested by Lightfoot (1989). The WRc (1996a) used linear logistic regression but also applied a generalised non-linear model and a generalised linear model without logistic transformation to the data. However, the WRc (1996a) had little success in fitting the generalised non-linear model to the data and inherent limitations were observed when attempting to fit the generalised linear model.

Limited resources meant that the survey was restricted to 1276 subjects, of which it was only possible to examine 585 for the health risk assessment. Most other studies obtained much higher survey numbers to calculate their results, for example Balarajan (1992, 1993) had a response rate of 7038 and 6875 for his prospective cohort studies. Cabelli (*et al.*, 1982) interviewed in excess of 10 000 swimmers over a six year study, Lightfoot formed her conclusion from analysis of 8420 respondents and Von Schirnding (1993) interviewed 5551 participants. Further work is required to accurately identify indicators representative of faecally polluted water and health risk to define a threshold limit. To establish this it is imperative that a standardised protocol and methodology for water quality sampling are developed. Until this is achieved the debate over selection of appropriate indicators will surely continue.

Chapter 6(c) Results and Discussion

6.c LITTER ANALYSIS

6.c.1 Introduction

Field experiments to assess how the public viewed three different compositions of debris were conducted at Whitmore Bay (Williams and Nelson, 1997). The categories were general litter (Group A), sewage related debris (Group B) and a combination of the two (Group C). see Plates 5.3.1 and 5.3.2. These were placed in a grid system on the beach and debris corresponding to the three categories were placed in grid cells in increasing quantities (refer to Methods, Section 6.3.1). The public were asked to assess at which grid point the debris would be objectionable. Data was also collected from the participants regarding the main attributes attracting them to Whitmore Bay and personal information, including sex, age, socio-economic status and whether or not they were locals or visitors (travelling in excess of 15 km to the beach).

In addition to the perception survey on the beach, a similar experiment was carried out at the University of Glamorgan laboratory, using photographic plates of the litter grids. Photographic plates or slides have been used by many researchers to investigate visual perception of landscapes and pollution (Daniel, 1976; Herzgog, 1985; Williams and Lavelle, 1990; House, 1995). This approach has been selected to overcome problems working *in situ*, but creating a controlled environment. House and Herring (1995) and Coughlin (1976) stated that by using slides the subjects are not biased by changing environmental conditions. Williams and Lavelle (1990) and Herzgog (1985) also found the usage of colour slides to be feasible with respect to evaluations of landscapes compared to real life. A sample of final year undergraduate students from the BSc Environmental Pollution Science course were used for the experiment. The objective was to investigate whether a relationship existed between the two sets of results, comparing the perception of litter *in situ* and photographs used in the laboratory. Use of photographic plates enable experiments to be conducted more easily using a laboratory, under controlled conditions. Also the logistics and budget for carrying out laboratory experiments are significantly less than working in the field. It was noted that the students had thorough knowledge of environmental issues, including beach litter. This was a pilot study.

The beach field work was performed over 5 days through hot weather in August 1995. Whitmore Bay is mechanically cleansed early each morning, therefore most of the debris deposited at the end of the day is predominantly visitor borne. During the perception survey, litter was recorded each evening to examine the volume and composition of debris deposited by beach users. Data was collected utilising three quadrats placed between the intertidal zone (using random numbers for positioning), and a 5m wide beach trawl along the strandline, following the procedure formulated by the Norwich Union Coastwatch Study (Rees and Pond, 1994).

6.c.2 Perception Grid Analysis *In situ* at Whitmore Bay

Field work was completed over the 5^{th}- 16^{th} of August 1995. One hundred and sixteen respondents were involved, selected from people passing the interviewers. Although it would have been more appropriate to take a random sample of beach users, it was impractical for people seated on the beach to leave their possessions to participate in the survey. To reduce bias a systematic approach was taken obtaining approximately equal numbers of males and females and also a balanced age range. Mean values were used as a gauge to contrast relationships between perception of gender, age and socio-economic classes. Overall mean grid perception values were 2.2 for rows A and B (with a standard deviation of 1.0) and 1.9 for row C (with a standard deviation of 0.97). All data failed the Kolmogorov-Smirnov normality test ($P<0.05$) and were consequently subjected to non-parametric analysis in the form of the Mann Whitney Rank Sum test (Jandel Scientific, 1995; Porkess, 1988; Siegel, 1956)

6.c.2.1 Gender Against Perception of Beach Debris

For both genders, the lowest tolerance to beach debris was category C i.e. a combination of categories A and B (Fig. 6.c.1). Females were more sensitive to accumulations of all three debris types. Findings by Morgan *et al.,* (1995) also found females to place higher priorities on a clean beach environment. Average values found for males was 2.0 and for females, 1.7 with a standard deviation (s.d.) of 1.0 and 0.95 respectively. The Mann Whitney Rank Sum test did not show a statistical difference across A or B for gender. However, for category C the differences in the median values among the two groups were greater than would be expected by chance, indicating there to be a statistical difference ($P < 0.05$) (Jandel Scienfific, 1995).

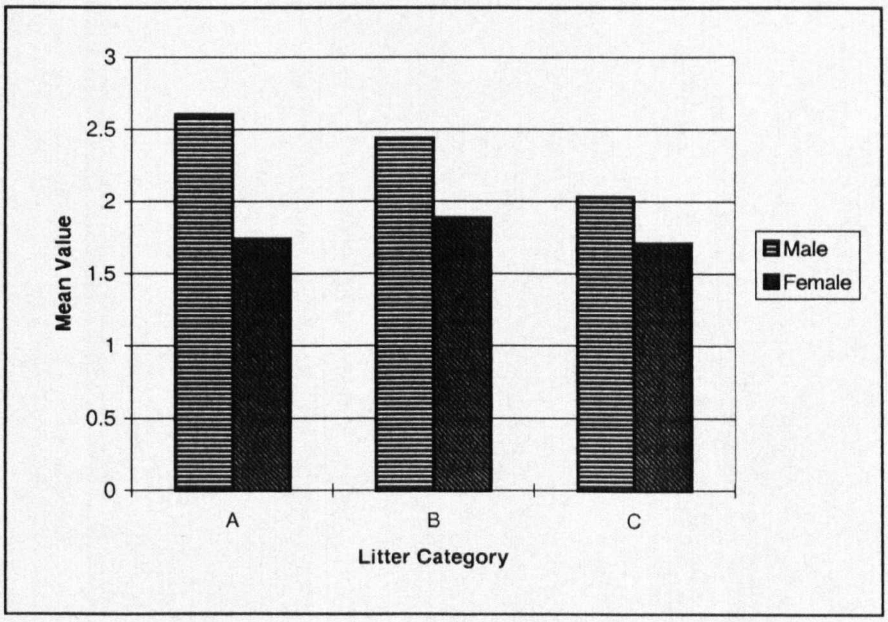

Figure 6.c.1 Gender vs. Litter Perception

6.c.2.2 Occupation Against Perception of Beach Debris

Table 6.c.1 lists occupation against the respective mean counts for each group against litter categories A, B and C. Occupation status was derived from the Government Statistical Service (1991). Comparison of results between occupations and debris classifications was interesting. Category C in all cases except for the professional group, was perceived to be the most obtrusive group of debris. In a few cases it was equalled, twice by category B and once by category A. No obvious difference emerged between groups A and B. Group 9 representing the retired section of the sample, although perhaps not significantly valid due to the low number of respondents, showed the highest resilience to perception of beach debris in all three cases. Another set pattern came out of group 4, representing skilled non-manual labour, which showed the highest sensitivity to beach debris across all three cases. The Mann Whitney Rank Sum test was applied to the Occupation data, which showed that although trends can be observed, the differences in median values among the socio-economic groups were not great enough to exclude the possibility that the difference is due to random sampling variability, indicating there is not a statistically significant difference ($P > 0.05$) (Jandel Scientific, 1995).

CLASS	OCCUPATION		GRID		
		N	A	B	C
1	Professional Occupations	18	2.11	2.16	2.27
2	Managerial Occupations	13	2.15	2.07	1.46
3	Skilled Manual Occupations	8	2.37	2.5	2.0
4	Skilled non - Manual Occupations	16	2.06	1.75	1.43
5	Partly Skilled Occupations	5	2.4	2.6	2.4
6	Unskilled Occupations	4	2.75	2.5	2.5
7	School	14	2.21	2.07	2.07
8	Housewife / Unemployed	27	2.11	2.22	1.77
9	Retired	9	2.88	2.77	2.11
10	Widowed	2	2.0	2.0	2.0

Table 6.c.1 Occupation vs. Debris Perception

6.c.2.3 Age Against Perception of Beach Debris

Table 6.c.2 highlights the relatively balanced range of ages and displays the mean value recorded for each litter category. People under 10 years of age were excluded from the study, therefore Group 1 comprised all respondents aged between 10-19. The upper two age categories were collapsed due to limited numbers forming Group 5, which comprised older people over 50. Debris in category C, comprising mixed debris, again proved to be the most obtrusive form of composition, except in the youngest age range where sewage related debris was calculated to be the most visually offensive (Table 6.c.2). The lowest age range, 10-19, scored lowest across both groups A and B, indicating the highest sensitivity over general and sewage-related debris. Group 3, aged between 30-39, were the most sensitive to category C and scored the second lowest to the 0-19 group for categories A and B. The highest tolerance to beach debris in categories A and B were in the 20-29 age range and 50+ for category C. These trends are not supported statistically by the Mann Whitney Rank Sum test at the P=0.05 level (Jandel Scientific, 1995).

AGE	GROUP	*N*	Grid A	Grid B	Grid C
10 - 19	1	25	2.0	1.88	2.0
20 - 29	2	25	2.44	2.44	1.92
30 - 39	3	25	2.2	2.16	1.76
40 - 49	4	21	2.24	2.38	1.81
50 +	5	20	2.3	2.3	2.05

Table 6.c.2 Age vs. Litter Perception

6.c.2.4 Comparison of Locals to Visitors Against Perception of Beach Debris

A distance of 15km was believed a reasonable radius to differentiate between locals and non locals. The data set consisted of 50 locals and 66 non locals. These numbers were significantly large to give a good statistical representation of the public's perception of the debris groups. Not surprisingly debris category C again came out as the most obtrusive collection of items, locals had an average grid perception value of 1.6, standard deviation 0.75 and visitors an average grid perception value of 2.1 standard deviation 0.96 (Figure 6.c.2). These results were supported statistically by the Mann Whitney Rank Sum test at the P=0.05 level (Jandel Scientific, 1995). In general the visitor group proved to have the most resilience to higher quantities of beach debris. Perhaps their understanding of beach debris was not as high and expectations of the visit experience might go well beyond the confines relating purely to beach quality.

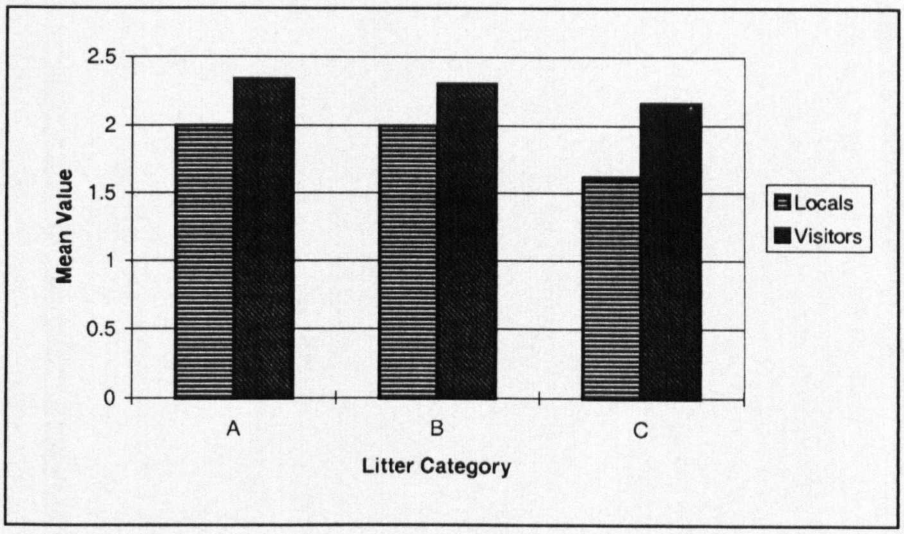

Figure 6.c.2 Visitor Type vs. Litter Perception

6.c.2.5 Attraction of Whitmore Bay

The beach users involved in the grid analysis were asked to select the most important aspects of a beach from five attributes. These attributes included Facilities, Access and Parking, Water Quality, Water Safety, and Views and Landscape. Water Quality was recorded to be the most important factor when selecting a beach, 78%, followed by Water Safety, 59% and Facilities, 38%. 'Access and Parking' and 'Views and Landscape' scored low. These results were verified using Chi-square analysis at the

P=0.05 level (Jandel Scientific, 1995). Contradictory findings were produced from the 1995 survey questionnaire, where only 7% thought the water quality at Whitmore Bay to be clean (see Section 6.d.3.1). It is also ironic with the history of poor water quality at Whitmore Bay that water quality should be the most significant factor in beach choice. There is a possibility that choice of water quality was influenced by media attention at the time, given to the pollution incidents at Oxwich Bay (The Independent, 1994; Wales on Sunday, 1994a). Water Safety, which also scored highly, was probably due to the large number of families and children present at the beach and Facilities probably scored highly for the same reason.

6.c.2.6 Perception Grid Analysis Using Photographs

Forty one students were involved in the laboratory study. Students were presented with photographs of all the litter grids used at the beach and required to view each individually for approximately 10 seconds, similar to the time period used *in situ* at Whitmore Bay. Table 6.c.3 details the average scores against gender. There was not a significant difference in tolerance between gender among students, identified by the Mann Whitney Rank Sum test (P=0.05; Jandel Scientific, 1995), compared to beach users, where females were more sensitive to litter. In contrast to beach users who selected group C to be the most visually offensive category, a combination of sewage-related debris and general litter, the students selected category B, sewage-related debris. In general, male students were more sensitive to litter than their beach user counterparts, but there was not a significant difference between female students and female beach users. The students mostly fell into the same age band and therefore precluded analysis of litter score against age. Results suggested that the students who are considered to be expert are not representative of the beach going population in opposition to findings by Williams and Lavelle (1990). Although their work was on landscapes and not pollution, they found experts to be representative of the general population. It might be suggested that students who selected sewage-related debris as the most offensive group of litter are more aware of the dangers inherent within that category of litter. Concurrent with these results, House and Herring (1995) also found sewage-related debris to be a significant factor in affecting the aesthetic quality of waterscapes, in agreement with Dinius (1981) who also noted the presence of litter to heavily influence the perception of pollution.

Gender	n	A	B	C
Male	26	2.23	1.77	1.81
Female	15	2.53	1.73	1.87

Table 6.c.3 Average Students Litter Score vs. Gender

6.c.3 Strandline/Quadrat Data

Table 6.c.4 summarises the strandline data and Table 6.c.5 displays data recorded in the quadrats over the survey days. Both collection methods show very high counts of polystyrene and plastics, which were the most prominent forms of debris discarded by visitors to Whitmore Bay. Other items with high counts were cigarette butts and miscellaneous items of paper, but both of these categories comprise mostly small items. Figure 6.c.4 graphs the main litter groups found on the beach over the whole survey and Figure 6.c.5 details a breakdown of the plastics group, which primarily consists of food/drink containers. Excluding the miscellaneous plastics group, plastic bottles proved the most recorded items (Table 6.c.4). It was noteworthy to find no glass present on the beach, a reflection in the volume of plastic used in society. Quadrats failed to a give true indication of the density of beach debris, highlighted by the low counts on 16 August, the day with the highest strandline debris load.

Litter	5 Aug	7 Aug	8 Aug	9 Aug	14 Aug	16 Aug	Total
Cans	2	6	7	16	4	30	65
Children fish netting	3	0	0	4	2	5	14
Crisp packets	26	45	24	37	31	33	196
Edibles	11	23	5	13	7	11	70
Butts	15	98	92	208	37	161	611
Cigarette packets	0	0	2	4	0	1	7
Food wrappers	32	16	68	23	39	57	255
Lolly/ice cream wrap.	7	43	2	16	6	18	92
Misc. plastics	20	25	19	17	24	15	120
Nappies	0	1	1	0	0	1	3
Paper	54	73	54	88	63	124	456
Plastic bags	3	14	3	3	5	7	35
Plastic bottles	7	28	12	19	12	14	92
Plastic cups	0	12	13	7	0	19	51
Plastic wrappers	0	13	7	19	1	14	54
Polystyrene items	38	56	126	122	40	211	593
Straws	2	3	3	14	2	4	28

Table 6.c.4 Strandline Data Record

	Quadrat 1	Quadrat 2	Quadrat 3
5 Aug	Cans 2	Butts 1	Butts 6
	Lolly sticks 3	Papers 3	Lolly stick 1
	Fish net 1	Polystyrene cont. 2	Papers 6
	Polystyrene cont. 2		Chip packet 1
7 Aug	Cans 1	Crips packet 1	Chip packets 3
	Food wrap. 2	Butts 4	Crisp packets 3
	Lolly sticks 2		Butts 3
	Lolly wrap. 3		Straws 2
	Misc.plastics 2		
8 Aug	Misc.plastic 1	Cans 1	Polystyrene cont. 1
	Papers 3	Chip packet 1	Crisp packets 2
	Polystyrene cont. 1	Clothing item 1	Sweet wrap. 2
		Lolly stick 1	Chip packet 1
		Lolly wrap.1	
		Paper 1	
9 Aug	Netting 1	Cans 1	Cans 2
	Papers 5	Crisp packets 4	Chip packets 2
	Poystyrene cont. 2	Butts 2	Butts 6
	Can 1	Papers 3	Food wrap.3
			Paper 1
14 Aug	Misc.plastic 1	Can 1	Chip
	Papers 3	Chip packet 1	
	Polystyrene cont. 1	Clothing item 1	
		Lolly stick 1	
		Lolly wrap. 1	
		Paper 1	
16 Aug	Food wrap. 1	Butts 3	String 1
	Lolly sticks 2	Polystyrene cont. 1	
	Polystyrene cont. 3	Straw 1	

Table 6.c.5 Quadrat Data Record

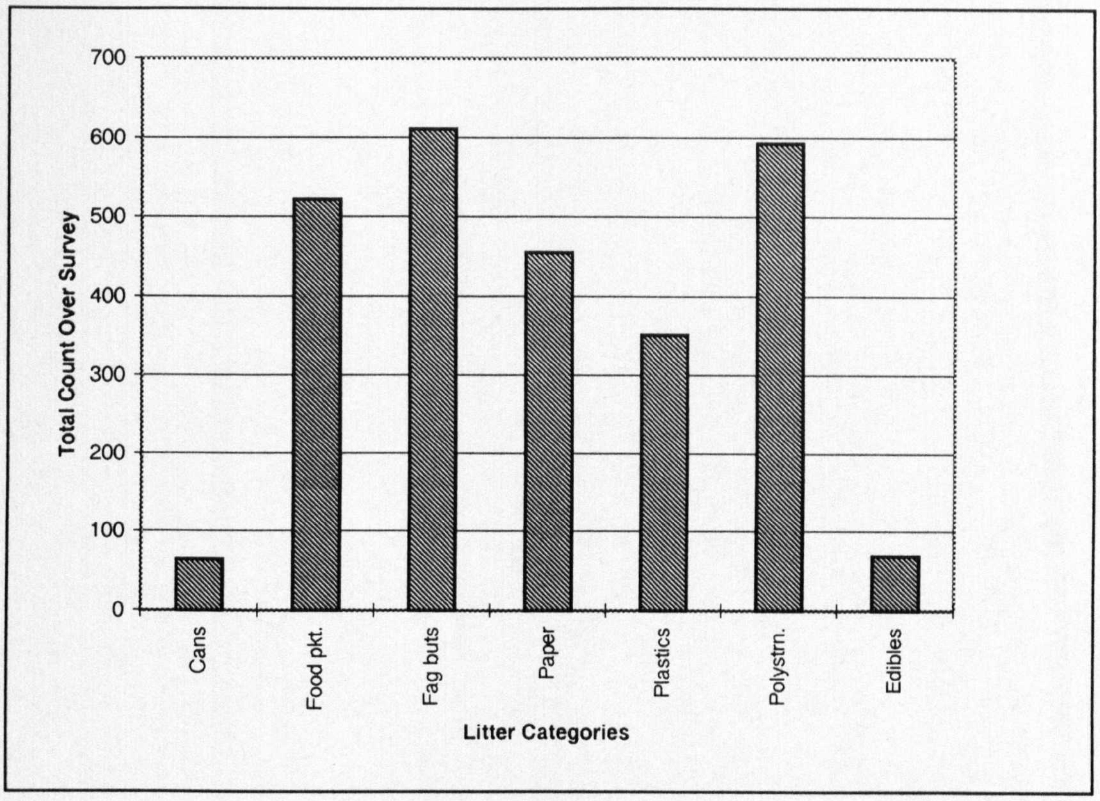

Figure 6.c.3 Number of Litter Items

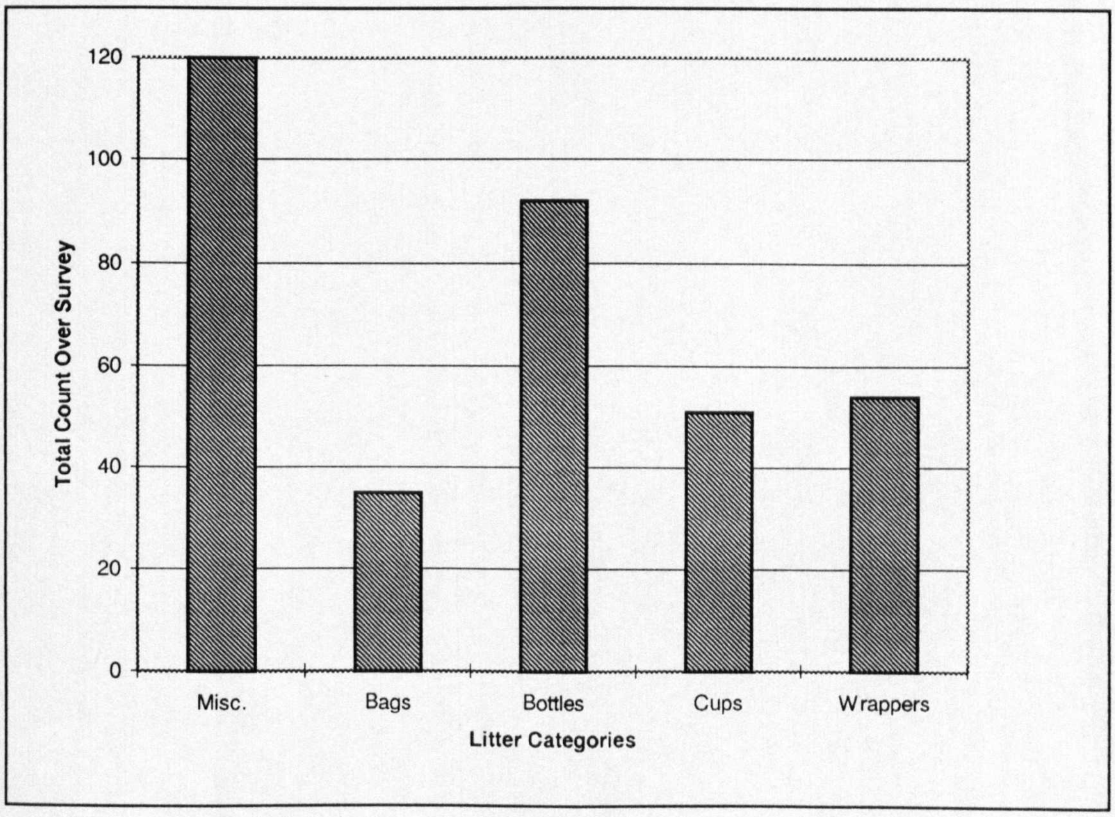

Figure 6.c.4 Number of Plastics Items

6.c.4 Summary of Litter Results

The enjoyment of a visit to the beach is affected by debris. Respondents gave an indication from the grid analysis a point at which debris density became visually offensive to the extent that they would refrain them from future visits. Results of the litter grid analysis suggest that the general public are more affected by a mixture of generic debris categories as opposed to the individual categories. The generic groups defined in this study were general items from visitor discards and sewage related debris, such as condoms and sanitary towel backing strips. The results showed females to be more sensitive to the perception of beach debris and females were also more perturbed by sewage contaminants than males. This may well be due to a higher recognition of these particular items. The concept of aesthetic pollution and public perception of coastal debris in aesthetic value has been addressed by House and Herring (1995) who focused on sewage derived debris. Their findings were at variance with the results of this research, concluding sewage derived contaminants to have a greater impact on the enjoyment of a visit to a beach than any other aesthetic pollution parameter.

Use of students, who represented experts, to appraise the quality of beaches using photographs proved to vary significantly from the *in situ* beach user surveys, in contrast to findings by House and Herring (1995), Williams and Lavelle (1990) and Coughlin (1976). However, supporting the lack of association between results obtained using photographs compared to the field work results, Turner (1977) stated that he is unconvinced that 2D visual stimuli are an acceptable surrogate for landscape.

Data recorded from the quadrat and strandline surveys showed visitor discards to be a major input of litter during the summer holidays. Few sewage-related debris items were noted, but plastics and polystyrene, both of which are very persistent materials, were prominent components of the litter observed. High volumes of strand line debris were found each day, which would definitely prove to be a very significant problem if not mechanically cleansed on a daily basis.

6.c.5 Discussion of Litter Results

The management of beach litter is a complex problem which requires an holistic view to find effective solutions. It goes beyond a national issue, noted by Grant and Jickells (1995) who commented on its international scope. Marine debris potentially effects the health of beach users, wildlife and aesthetic quality of the landscape (refer Chapter 4, Section 4.a.1). Environmental management, which encompasses beach quality needs to appreciate both the ecological aspects of the coast and address the dimensions of sustainable management planning. Any strategy must consider the carrying capacity and fragility of the local environment and the activities, expectations and recreational experience of the beach user. Failure to do so will ultimately lead to a degraded coastal environment which ultimately will lead to loss in tourism income (Fanshaw, 1996; Phillip, 1997; Williams and Davies, in press).

Beach Debris originates from four main sources, visitor discards mentioned above, marine debris, estuarine and riverine pathways including combined sewer outflows and the sewerage system (Williams and Nelson, 1997). The debris composition in the strandline and quadrat analysis suggest that the major input at Whitmore Bay is through visitors. Litter incurred directly as a result of tourism is site specific and limited (Scott, 1972). The long term resolution to managing beach litter is to tackle the problem at source (Earll *et al.,* 1997). However, the mobile nature of litter in the aquatic environment make it very difficult to source, making it an international problem. The Norwich Union Coastwatch study identified coastal litter on the British coast stemming from 27 other countries (Rees and Pond, 1997). International protocols are in place such as the MARPOL Convention (1973/78). However, although good in theory, policing marine craft is an almost impossible task (Grant and Jickells, 1995). Further compounding the issue of litter management is the lack of accountability of the manufacturers, who do not burden the responsibility or cost of clearance, which is heavy. For example, the estimated cost of clearing two beaches in Weston-Super-Mare exceeded £100,000 during 1994 (Acland, 1995). High cost of clearance is not just restricted to the UK, for example Olin *et al.,* (1995) claimed that beach cleansing the Bohuslan coast during 1993 totalled £937,000. Therefore, it is not practical to apply the

'Polluter Pay' principle to litter management. Currently no answer exists to retrospectively sourcing litter back to its origin, made more complicated through litter travelling inter-country. In addition there is no simple technological fix to dealing with sewage-related debris (Simmons and Williams, 1994), which has an extreme detrimental aesthetic impact on the coastline (House and Herring, 1995). One long term solution tackling the problem at source would be to encourage better material design, allowing debris to quickly break down organically.

Before any major action plan to tackle debris on the British coastline can be initiated the problem needs to objectively quantified. The most important step is to be able to accurately measure litter before management strategies can be formed. Various initiatives have been set, for example the Norwich Union Coastwatch Study (Rees and Pond, 1994), Marine Conservation Beachwatch programme (MCS, 1997) and the Thamesclean Project (Lloyd, 1996). All programmes have been designed with their own agenda, the result is incompatibility. A new framework has been developed (The ABCD model) to standardise monitoring of litter. A group of experts which designed the scheme include members from the Environment Agency, TBG and academics, forming the National Aquatic Litter Group (NALG; Earll and Jowett, 1998).

The NALG (Earll and Jowett, 1998) is providing a positive advance in attempting to standardise measuring and monitoring methods to build a complete national profile of the distribution and composition of beach litter. Through pooling of data it will be possible to build a comprehensive database to analyse specific materials, mobility and persistence of marine debris and isolate hot spots. Earll (1996) has suggested establishing litter species to further categorise litter, which will aid in communication of material information and link measuring techniques to management, an essential step dealing with coastal pollution. These mechanisms which facilitate measuring of litter will provide a data base and platform to underpin change in the way litter is managed.

To move forward public pressure is necessary to change attitudes and opinions. Environmental groups such as the Women's' Environment Network and Surfers Against Sewage are trying to encourage manufacturers to change their advice on disposal of feminine hygiene items, for example. Cultural change is necessary to move from a 'flush it down the toilet' approach to a 'bag it and bin it' approach, used in most of Europe. Grant and Jickells (1995) advocate public pressure to deal with discharging of pollution at sea by providing waste storage facilities at sea and waste disposal facilities at ports. These measures must be supported by strict legislation to be effective (Simmons and Williams, 1992); voluntary agreements do not work well, for example the MARPOL Convention (1973/1978) has had little impact. The industrial community will only be motivated to act if faced with prosecution. It is also down to the individual to take greater responsibility. The EC Charter on Environment and Health (WHO, 1989a p.3) states that 'every individual has a responsibility to contribute to the protection of the environment, in the interests of his or her own health and that of others'. A government white paper (Department of Health, 1992) supports this view by recognising the relationship between the quality of the environment and health consequences.

In the interim period, education of the public is essential, encouraging correct disposal of items, for example female hygiene objects (Williams and Nelson, 1997); a view shared by Fuller (1993) who stated that managers should concentrate on education of the public. A typical example is the 'Bag it and Bin it' slogan, which will hopefully reduce the load on the sewerage system. All too often in coastal management issues the views of the consumer are bypassed by the decision makers. Morgan *et al.* (1993) stated that beach management must consider the perception of the beach user for effecting sustainable management strategies. Wales will see a difference in the sewerage system, with new developments already taking place to improve the processing of waste (Welsh Water, 1996a). However, the problem of CSOs will continue to cause problems well in to next decade. Although technology exists to remove solids from CSOs, there are over 2500 in Wales, all of which need to be re-mapped (Welsh Water, 1996a). In terms of sewage-related debris on the coast recent UK legislation placing a requirement of 6mm mesh wire screens on all shore-based sewage outlets should create aesthetic improvements (Phillip *et al.,* 1997).

With respect to the three beaches studied all three respective Councils are addressing the problem of beach debris during the summer months by mechanically raking the beaches.

Even though this is curative rather than preventative, failure to keep the beaches clean will undoubtedly lead to reduced visitor loads, on which local economies are often heavily dependent upon in terms of revenue. Results from this study showed that a clean beach is a major factor in attracting tourists. In addition to continued education programmes discussed, regular maintenance of litter bins would be recommended (Williams and Nelson, 1997). A clean environment encourages users to behave in an appropriate manner. To meet long term objectives of tackling coastal debris effective sourcing of litter must be accomplished, aided by the use of photographic logs (Earll and Jowett, 1998).

The novel approach used for the grid analysis at Whitmore Bay gave valuable information regarding perception of beach litter. Results highlighted the negative implications of beach debris, and acknowledgement of the perception to beach pollution is essential in managing the coastline and beaches. More research into hazardous items on beaches should be carried out. This view was echoed by the WHO (1994a) who recognised the need for more work in assessing the association between health risks and both infectious and man made environmental hazards. Further work is suggested increasing the number of litter objects used for the grid analysis, incorporating additional quadrats to the survey work to achieve more representative results and expanding the number of beaches investigated. With respect to the use of photographic plates to gauge perception of litter it is suggested the inclusion of both a sample of beach users in the analysis of visual props and a larger sample group of experts covering a range of ages and socio-economic status to ensure a more comparable study.

Currently no standardised litter survey exists. Since the beach surveys a comprehensive litter survey design has been developed by the NALG (Earll and Jowett, 1998) and is now operational (refer to Chapter 4, Section 4.a.6.1). Further studies should be based around this methodology for comparison of data, to help in the understanding of litter trends and ways to manage coastal debris in the future. In a regional context pro-active developments in the form of the Green Sea Initiative, Coastal Forum in Wales and the Severn Estuary Strategy should have a positive impact on the health of Welsh beaches, which will hopefully benefit the tourism industry within the Principality.

Chapter 6(d) Results and Discussion

6.d QUESTIONNAIRE SURVEY 1995

6.d.1 Survey Response

The 1995 questionnaire survey was carried out over six days during August, including the 7, 8, 9, 14, 15 and 16. A total of 1276 completed surveys were obtained using questionnaires A (QA) and B (QB) (Appendix I), and analysed (refer to Methods, Section 6.4.8). Table 6.d.1 summarises the 1995 survey data response rate per day and Table 6.d.2 summarises the total telephone survey response numbers. The data is broken down in more detail in relevant sections. QA (1038 responses) was aimed at respondents over the age of 10, and included a request for information on their beach activities, health, foods eaten, perception of beach aspects and personal details. QB (238 responses), requested information on children under 10, and was filled in by their parents, who were also requested to answer questions on their perception of beach aspects. The information on the children included their activities, health and foods eaten for the health risk study.

Both questionnaires requested the respondent to give their telephone numbers for a follow up post beach interview, using a telephone questionnaire QT (Appendix I), to investigate the rate of illness experienced within 10 days of being to the beach (see Section 6.b.1). A total of 593 gave their telephone number, of which 585 were contacted. Data collected from questionnaires A and B were used in the epidemiological-microbiological health risk study. Problems occurred obtaining perception data from QB, with parents frequently leaving out information on themselves, once they had finished the section on their children. Therefore the perception data was based primarily on QA, which provided a representative sample of beach users during the survey days. Table

Public Perception and Coastal Pollution at Identified Beaches in South Wales.

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6.d.3 summarises the environmental conditions during the field work and positions of the tide.

	7 Aug	8 Aug	9 Aug	14 Aug	15 Aug	16 Aug	Total
QA	162	157	232	195	144	148	1038
QB	0	31	61	20	89	37	238
Total (n)							**1276**

Table 6.d.1 Questionnaire Response per Day 1995

Telephone Questionnaire	Response
QA Not Ill	383
QB Not Ill	105
QT Ill	97
Total (n)	**585**

Table 6.d.2 Telephone Response 1995

Date	H.Tide (Time)	Tide Height (m)	Max. Temp. °C	Rainfall (mm)	Hours Sunshine per day
7 Aug	1524	13.0	22.6	0	11.2
8 Aug	16:34	13.8	21.2	0	13.8
9 Aug	17:34	14.6	22.6	0	13.7
14 Aug	21:27	15.0	21.3	0	8.9
15 Aug	22:03	14.4	23.7	0	12.8
16 Aug	22:33	13.7	26.0	0	12.6

Table 6.d.3 Environmental Conditions During Survey 1995

6.d.2 Demographic Characteristics

6.d.2.1 Age distribution

Table 6.d.4 describes the distribution of respondents aged over 10. Children under the age of 10 were believed to be too young to be involved in the perception survey. The majority of the sample were aged between 10 to 50, with the 30-40 category being the highest proportion of respondents (32%), followed by the 40-50 (24%). It is likely that this age band represented the high proportion of mothers on the beach. There was a roughly equal ratio of people aged between 10-20 and 20-30 (18%).

Age Category (yrs)	Frequency	Percentage %
10-19	179	18.1
20-29	181	18.3
30-39	312	31.2
40-59	238	24.0
>60	65	8.4
Missing Value	48	4.6
Total (n)	1038	100

Table 6.d.4 Distribution of Respondents Over Age 10

6.d.2.2 Gender distribution

A higher proportion of females were interviewed (72%), compared to males (23%), detailed in Table 6.d.5. There are a number of possible reasons for this. Firstly, from observation it became apparent that females were more willing to fill in the questionnaire, giving a higher response rate. And secondly, a large number of beach users were mothers, supporting the age theory above.

Gender	Frequency	Percentage %
Male	238	22.9
Female	743	71.6
Missing Value	57	5.5
Total (n)	1038	100

Table 6.d.5 Distribution of Gender

6.d.2.3 Socio-Economic Status

Composition analysis of socio-economic status (Table 6.d.6) revealed that 40% of those questioned were employed and 33% were housewives. Again the latter adds further weight to the high proportion of women and mothers on the beach. In addition 64% of those employed were women (see Chapter 7, Section 7.4).

Socio-economic Status	Frequency	Percentage %
Employed	392	37.7
House Wife	326	31.6
Student	150	14.4
Unemployed	46	4.3
Retired	62	6.0
Missing Value	62	6.0
Total (n)	1038	100

Table 6.d.6 Distribution of Socio-Economic Status

6.d.2.4 Geographical Distribution of Respondents

The highest proportion of visitors were day trippers, 61%, travelling over 10 miles to the beach (Table 6.d.7). Eighteen percent were locals and 21% on holiday. Observation of the telephone survey indicated the Newport code to be the most prominent, implying a large proportion of people from the Gwent region. Gwent and the Valleys provide the main source of day trippers to Barry Island. However, although Barry Island is the main catchment for Gwent, Porthcawl receives a large segment of day trippers from the western Valleys.

Origin	Frequency	Percentage %
Holiday	197	18.9
Travelled > 15km	597	57.5
Live locally	201	19.4
Missing Value	43	4.2
Total (n)	1038	100

Table 6.d.7 Distribution of Visitor Type

6.d.3 Perception of Beach Pollution

Particular questions asked in the questionnaire schedule offer the opportunity for the respondent to select more than one category, including Q7, Q10, Q15, Q20 and Q21. The percentage scores for each category are calculated as a ratio of the total sample, therefore, the total will not necessarily equate to 100.

6.d.3.1 Selection of Whitmore Bay and Reasons for Not Participating in Water Activities

The respondents in the study were asked to select the main reasons for choosing to visit Whitmore Bay, from a list of attributes detailed in Table 6.d.8. Locality was the prime reason with distance scoring 76%, probably due to the very high day tripper load (58%) (Table 7). Facilities also scored highly (40%), not surprisingly. One of the main attributes to Barry Island is the considerable amount of entertainment provided along the sea front, including funfair, amusements, crazy golf and plenty of food installments. Clean beach ranked in third (37%), the sand at Whitmore Bay is mechanically raked each day, making it a clean beach. Only 10% placed importance on the resort winning a seaside award, which is in contradiction to findings of a later question which found 70% of beach users claimed a beach award to be an important criteria when selecting a destination. Less than 7% stated clean water as a reason for visiting the beach. Research by Cutter *et al.* (1979) similarly suggested that beach users opt for a less than ideal environment in favour of other factors such as accessibility.

Reason	**Frequency**	**Percentage**
Distance	789	76.0
Facilities	415	40.0
Clean Beach	382	36.8
Cost	338	32.6
Safety	251	24.1
Scenery	179	17.2
Beach Award Flag	103	9.9
Clean Water	68	6.6
Suitable Conditions	47	4.5
Other	66	6.4

Table 6.d.8 Main Reasons for Visiting Whitmore Bay

The questionnaire asked respondents to give their reasons for not swimming at Whitmore Bay during the survey days (Table 6.d.9). Even though the water passed EC Bathing Water Directive standards (CEC, 1976a) 55% chose not to swim, believing the water to be dirty. This may be the result of the water being highly turbid (see Section 6.e.5.1), but also might have been due in part to bad publicity at the time of swimming related illness reported at Oxwich Bay, 50km away (The Independent, 1994; Wales on Sunday, 1994). Previous work by other authors has suggested that water clarity is an important factor in the public's participation in water activities (Burrows and House, 1989; Hertzgog, 1985; Ditton and Goodale, 1974). Cold water figured highly with 23% as a reason for not swimming. These results are in contrast to work done by the Robens Institute (1987) who found at two resorts in the South East of England that cold water accounted for 37% of their sample choosing not to swim and only 8% because they thought the water was dirty. During summer months the seawater temperature was reasonably warm, averaging 19C through August. Chi-square analysis was applied to the data to see if there were any associations between perception of the water quality and age, gender, socio-economic status and visitor status. There was no statistical evidence to support any significant association between the former three variables. However, day visitors travelling over 15 km (day trippers) to the beach were more likely to select water quality as the main reason for not swimming compared to locals and holiday makers (P=0.05).

Reason	Frequency	Percentage
WQDIRTY	570	55.0
COLD	242	23.3
HEALTH	86	8.3
CANTSWIM	74	7.1
DONTSWIM	66	6.4

Table 6.d.9 Reasons for Not Swimming

6.d.3.2 Concern over Pollution at Whitmore Bay and the South Wales Beaches

When asked if the participants had heard of beach pollution on British beaches, 86% stated that they had. Over 57% said they had been exposed to bad press over beaches through the media, mainly television, 22% through environmental groups, which covers a wide band with water quality being high on the agenda in South Wales and a further 10% from Surfers Against Sewage (Table 6.d.10). Respondents were also asked if this information worried them (Table 6.d.11). Out of the high response, 85% declared that they were concerned over information regarding pollution of beaches, in agreement with a survey by the Times (1991), which found beach pollution to rank second in public concern over pollution issues. In addition similar research conducted by the Robens Institute (1987) also found television to be the main conveyor of information regarding coastal pollution which worried over 77% of respondents involved in their survey. Chi-square analysis was used to investigate if there were any associations between information worry and socio-demographic characteristics (Table 6.d.12). There was no statistical evidence to support any significant association between gender, socio-economic status and visitor type. However, both age groups between 30-39 and 40-49 were more likely to express worry over pollution on beaches (P=0.05) (Table 6.d.13).

	Frequency	Percent
Environmental Group	223	21.5
Local Authority	164	15.8
Water Authority	117	11.3
Surfers Against Sewage	103	9.9
Media	589	56.7

Table 6.d.10 Information Source of Pollution on British Beaches

	Frequency	Percent
Worry	878	84.6
No Worry	54	5.2
Don't Know	47	4.5
Missing Value	59	5.7
Total Media	1038	100

Table 6.d.11 Worry Level over Pollution of British Beaches

	Chi-Square χ^2	DF	Significance
Age	21.80908	4	0.00022
Gender	1.71025	2	0.4253
SES	6.25623	4	0.18081
Visitor Type	8.25859	2	0.01609

Table 6.d.12 Chi-square - Level Association Between Worry Level and Socio-Demographic Characteristics

Age	No worry	Worry	Row Total
10-19	13	150	163 18.3
20-29	8	148	156 17.5
30-39	8	277	285 32.0
40-59	8	217	225 25.3
>60	10	52	62 7.0
Column Total	47 5.3	844 94.7	891 100.0

Table 6.d.13 Chi-square - Level Association Between Worry Level and Age

6.d.3.3 Public Perception of Coastal Pollution

Participants were asked to state whether they had been put off by pollution along the South Wales coastline. A higher proportion reported that they had been more concerned with sea pollution (45%) than pollution along the shore (34%) with 82% believing the sea around South Wales beaches to contain a degree of pollution (Table 6.d.14). The questionnaire provided room for respondents to highlight specific beaches that they believed to be of poor quality (Table 6.d.15). The two major resorts in South Wales were highlighted, Barry Island (8%) and Porthcawl (9%). A low proportion claimed Gower Beaches to be polluted (2%). This was probably related to media coverage at the time over two incidents of illness believed to have been related poor water quality at Oxwich Bay (see above).

	Frequency	Percentage
Sea pollution	469	45.2
Shore pollution	354	33.7

Table 6.d.14 Concern Over Sea and Shore Pollution

Barry Island	Gower	Porthcawl	Other Welsh beaches
7.8%	1.7%	8.8%	2.6%

Table 6.d.15 Concern Over Pollution at Welsh Beaches

With respect to perception of water quality *per se*, 30% of the sample group believed the sea at Whitmore Bay to be Very Dirty with a further 40% believing it to be Dirty, forming a total of 70% perceiving the water to be of poor quality (Table 6.d.16). Application of Chi-square analysis to the data (Table 6.d.17) proved females to be more sensitive to perception of water quality than males and also day trippers were more likely to perceive the water quality at Whitmore Bay to be dirty in contrast to holiday makers and locals (P=0.05). Previous studies on coastal pollution also found females to be more sensitive to pollution of beaches than their counterparts (Williams *et al.,* 1993; Morgan *et al.,* 1995; Williams and Nelson, 1997). No statistical difference between different ages or socio-economic status was evident (P = 0.05).

	Frequency	Percent
Very Dirty	306	29.5
Dirty	412	39.7
Okay	196	18.9
Clean	51	4.9
Very Clean	15	1.4
Don't Know	36	3.5
Missing Value	22	2.1
Total	1038	100

Table 6.d.16 Perception of Water Quality at Whitmore Bay

	Chi-Square χ^2	DF	Significance
Age	11.2378	9	0.25976
Gender	11.26342	4	0.02376
SES	13.88909	12	0.30785
Visitor Type	113.93645	12	=<0.001

Table 6.d.17 Chi-square - Level Association Between Perception of Water Quality and Socio-Demographic Characteristics

Table 6.d.18 relates to perception of the most offensive forms of sea pollution, not specific to Whitmore Bay. Floating objects (75.5%) were perceived as the most offensive forms of pollution in bathing waters, which included faeces, condoms and sanitary towels, followed by scum (66%) on the water, usually surfactants and not sewage. Research by David (1971) and Nicolson and Mace (1975) also showed floating objects to be figure prominently in being visually obtrusive to water recreationalists. Simmons and Williams (1994) described the increase in sewage-related debris along British shores, and highlighted the problem caused by their longevity and persistence (refer Chapter 4, 4.a.3). The third most offensive pollutant recorded was discoloured water (58%). The Robens Institute (1987) also found similar numbers (54%) in their survey quoting

discoloured water as figuring prominently in terms of visual pollution, supported by earlier work by Dinius (1981). Offensive smells (37%) were recorded as the next highest score, a factor noted by Moser (1984) as very offensive to water users. Table 6.d.19 relates to the most observed forms of pollution seen in the sea on the day of the surveys, at Whitmore Bay. As can be seen discoloured water traded place with floating objects, jumping from third to first place on the list with 77%, an increase of 20%, in contrast to results listed in Table 6.d.14. This is most likely due to the high turbidity from a heavy silt load in the Severn Estuary (Severn Estuary, 1997). There is a public mis-conception that discoloured water is an indicator of pollution, however, this is not necessarily true (Dinius, 1981; Smith, 1995a). Floating objects scored 37% at Whitmore Bay, which is almost half that recorded in Table 6.d.18, implying a low presence of floating objects in the seawater at the beach.

Pollution	Frequency	Percent
Floating objects	784	75.5
Scum	685	66.0
Discoloured Water	599	57.7
Smell	383	36.9
Oil	336	32.4
Other	77	7.4

Table 6.d.18 Perceived Most Offensive Forms of Sea Pollution

Observed item	Frequency	Percentage
Discoloured Water	800	77.1
Float objects	383	36.9
Foam	365	35.2
Smell	232	22.4
Film	229	22.1
Oily	176	17
Organic	116	11.2
Other	35	3.6

Table 6.d.19 Most Offensive Forms of Sea Pollution Observed at Whitmore Bay

The final question on perception of pollution required the respondents to identify which three forms of litter objects and sewage-related debris have been noticed along the South Wales coastline (Table 6.d.20). Food packaging was the most observed category both on the beach (77%) and in the sea (29%), along the South Wales coastline in agreement with the findings of the litter trawl survey (refer Chapter 4, Tables 6.c.4 and 6.c.5). Both aluminium cans and plastic bottles were recorded as being more prominent on the beach than the sea, even though plastic bottles were the most highly noted category viewed in the water (36%). Sewage-related debris obviously had higher counts for the sea, for example in the sea condoms scored 15% compared to 12% on the beach and sanitary items observed in the sea scored 13% in contrast to 10% on the beach. Faeces had high scores for both the beach (26%) and sea (17%), which is probably a combination of both sewage and animal waste. Oil and chemicals were also more prevalent in the sea than the beach. For oil the sea scored (9%) and the beach (13%) and for chemicals the sea had a very high count of (18%) and the beach (2.6%). Williams and Morgan (1995) and Young *et al.,* (1996) also found oil to be high on beach users concerns over pollution of beaches.

	Beach		Sea	
	Freq.	*%*	*Freq.*	*%*
Food packing	803	77.3	296	28.5
Ally cans	634	61.2	214	20.6
Plastic bottle	665	64.1	372	35.8
Faeces	266	25.6	181	17.4
Condoms	119	11.5	159	15.3
Sanitary item	101	9.7	139	13.4
Oil	90	8.7	134	12.9
Chemicals	27	2.6	185	17.8
Other	133	12.8	50	4.8

Table 6.d.20 Most Offensive Forms of Sea and Beach Pollution Observed on South Wales Coastline

6.d.3.4 Summary of Perception to Beach and Sea Pollution

In summary the general perception of the seawater at Whitmore Bay was that it was of poor quality. This was an underlying theme which consistently manifested itself through different questions, from figuring highly as a reason for not swimming to 70% believing the water to be dirty. It would appear that the major cause for the perception of beach users to believe the quality of the seawater to be poor is due to the high turbidity, with a proportion of respondents claiming the main pollutant at the Bay to be discoloured water. As mentioned earlier, turbidity is not necessarily an indication of poor water quality, but at Whitmore Bay the lack of clarity is due to a high sediment load from the Severn Estuary. In the context of socio-demographic characteristics, females and day trippers were more sensitive to coastal pollution, in particular poor water quality. Floating objects were perceived to be a the worst form of sea pollution, but under 50% noted this as a problem at Whitmore Bay. Food packaging items were the most observed items at the beach, both on shore and sea, which confirms similar findings from the litter analysis (Tables 6.c.4 and 6.c.5). For a discussion on aesthetic indicators see Section 6.e.6.

6.d.4 Results of Beach Flag Awards

The questions on perception of beach awards were ordered in such a way as to gauge the publics true knowledge of them, before predisposing them to the different systems in operation. The literature showed a dearth of research into perception of beach users to seaside award schemes. It has been suggested that there is a great deal of confusion over the different systems available (Hines *pers.comm.*, 1995; Nelson and Williams, Nelson *et al.*, in press (a)). Approximately half of the respondents claimed to be aware of the various beach award flags (49%) on the market (the questionnaire made clear that these systems did not relate to lifeguard patrol/danger flags). In a mini study conducted by Paul (*pers.comm.*, 1997) in Lime Regis > 72% failed to recognise either the Blue Flag or the TBG Seaside Awards. Lack of knowledge over beach flags might give reason to findings of an earlier question which showed respondents gave a low rating to beach flags as a preference criteria when selecting a beach, in relation to other attributes such as distance from home and cost of trip.

The respondents were given an open ended question, requiring them to state their understanding of what a flag represented at a beach (not necessarily Whitmore Bay). The intention was to ascertain how the public identify with different types of flags on beaches, not specifically beach awards. Table 6.d.21 shows the level of understanding towards a flag displayed at a beach in terms of the total population and as a percentage of those that responded to the question. On the assumption that the limited response rate is due to lack of understanding, the following discussion is based on the total sample. Of the total sample 14% believed a flag represented cleanliness with a further 4% believing a flag to mean the beach was clean and safe and 19% claimed to associate a flag with a beach award scheme. Just over 20% of the sample believed a flag to either signify safety (6%) or danger (11%). Almost one third (30%) had no understanding of beach awards at all. House and Herring (1995) found that 22% of their sample thought beach award flags represented bathing areas that were safe to swim in, compared to 6% of respondents in this study who believed a flag represented safety.

Meaning	Frequency	Percent of Total Sample	Percent of Respondents
Clean	140	13.5	20.0
Beach Flag	130	12.5	18.6
Danger	118	11.4	16.9
Safety	60	5.8	8.6
Clean and Safe	42	4.0	6.0
No Understanding	210	30.0	30.0
Missing Value	338	32.6	--
Total (n)	1038	100	700

Table 6.d.21 Understanding of a Beach Flying a Flag

A higher response rate was achieved when asking how important a beach award flag was when selecting a beach (89%), compared to the previous question. Again the results are discussed in terms of the total sample, Table 6.d.22, which also shows the percentage response as a ratio of those that answered the question. A total of 72% believed the attainment of a beach award flag to be important in influencing their selection of a beach, split between very important 45%, and vaguely important 27%. Chi-square (χ^2) analysis was used to investigate associations between influence of beach award flag and age, gender, socio-economic status and visitor type were investigated, summarised below (Table 6.d.23). No statistically significant association existed between influence of a beach award flag and gender or visitor type. However, the oldest age range between 40->60 years of age placed more importance on beaches attaining a beach award, followed by the 30-39 age category (Table 6.d.24). Very little importance was shown by the younger groups. Also employed persons and housewives placed more importance on a beach award flag in beach choice than students, retired or unemployed people (Table 6.d.25). The results of this question conflict with the response from an earlier question which showed 50% of the sample population to be unaware of beach awards and yet in this question 72% reported that they played an important role influencing beach selection. There are no obvious reasons for this discrepancy.

	Frequency	Percent of Total Sample	Percent of Respondents
Important	412	39.7	44.6
Vaguely Important	245	23.6	26.5
Not Important	118	11.4	12.8
Undecided	148	14.3	16.0
Missing Value	115	11.1	--
Total (n)	1038	100	923

Table 6.d.22 Influence of Beach Award on Beach Selection

	Chi-Square χ^2	DF	Significance
Age	33.92281	9	0.00009
Gender	3.8532	3	0.27776
SES	27.41273	12	0.00674
Visitor Type	14.42359	14	0.41865

Table 6.d.23 Chi-square - Level of Association Between Influence of a Beach Award and Socio-Demographic Characteristics

Age	Important	Vaguely Important	Not Important	Undecided	Row Total
10-19	64	56	25	33	178 19.4
20-29	56	45	26	37	164 17.9
30-39	133	89	31	45	298 32.5
40>60	155	54	36	33	278 30.3
Column Total	408 44.4	244 26.6	118 12.9	148 16.1	918 100.0

Table 6.d.24 Chi-square - Level of Association Between Influence of Beach Flags and Age

SES	Important	Vaguely Important	Not Important	Undecided	Row Total
Employed	152	107	52	47	358 39.5
H-Wife	143	72	32	57	304 33.5
Student	50	43	23	27	143 15.8
Unemployed	19	8	7	9	43 4.7
Retired	39	11	4	5	59 6.5
Column Total	403 44.4	241 26.6	118 13.0	145 16.0	907 100.0

Table 6.d.25 Chi-square - Level of Association Between Influence of Beach Flags and SES

When asked if Whitmore possessed a beach award flag, the majority of participants in the study (53%) reported that they were unsure (Table 6.d.26). Again this gives weight to the confusion over beach flags. It is not believed the questions are ambiguous or misleading.

	Frequency	Percent of Total Sample	Percent of Respondents
Yes	114	10.9	12.3
No	264	25.4	28.4
Don't Know	549	52.9	59.2
Missing Value	111	10.8	--
Total (n)	1038	100	927

Table 6.d.26 Knowledge of Flag Status at Whitmore Bay

Three main beach awards were available in the UK in 1995 (Chapter 4, Section 4.a.7), the FEEE Blue Flag (1997) and the Tidy Britain Seaside Awards (TBG, 1995) which formed two categories. The Premier Seaside Award required water quality at beaches to reach EC Guideline standards and the Seaside Award which required the water quality to reach EC Mandatory standards. Both of these two awards were further stratified to resort and rural beaches. The main difference being the necessity of a resort beach to offer facilities such as toilets, café and close proximity to urban areas.

Respondents were asked to state whether they understood the meaning of the Blue Flag and Seaside Awards. Only 35% thought they understood the Blue Flag and 21% thought they understood the Seaside Awards (Table 6.d.27). There were a significantly higher proportion of respondents who believed they understood the meaning of the Blue Flag compared to the Seaside Award Flags, verified using chi-square ($P<0.05$)

Values in %	Understand	Don't understand
Blue flag	34.9	65.1
Seaside award	21.2	78.8

Table 6.d.27 Level of Understanding of Beach Award Flags at Whitmore Bay

The final question in the 1995 beach award sequence investigated the accuracy of respondents knowledge and level of understanding in identifying the FEEE Blue Flag and TBG Premier and Seaside Award with a set of criteria (Table 28). Clean beach, clean water quality (EC Guideline standard), safety (in the form of lifeguard provision), toilets, popular (tourist), and dog control apply to the Blue Flag and both Premier Seaside Award and Seaside Award (resort beaches). The rural categories for the TBG Awards differ in that for water quality only the EC Mandatory standard is required, safety provision includes equipment, not necessarily lifeguard, the beach does not have to have toilets or state dog control. For a full list of attributes for the awards (see Appendix IV). On average over 70% of respondents refrained from answering this question, the low response implying a lack of knowledge. On this premise the results detailed in Table 6.d.14 are percentage scores of the total sample. In addition is was impossible to calculate percentage scores as a ratio of the response rate for each variable, as the response rates were different. This would have introduced bias in the results. For example only 21% of the total sample responded to the boating attribute in contrast to 43% response to the clean beach attribute.

Two of the criteria, namely sandy beach and boating facilities were used as dummy variables, not related to any of the beach award flags. Both received low response rates from the sample group, indicating that these were not clearly identified with any flags. Respondents identified more accurately criteria attributed to the Blue Flag than the TBG Seaside Awards, reflected by higher percentage scores for the clean beach, water quality and safety criteria. Toilets were next on the list, followed by the beaches being popular, which represented a tourist beach, also specified under conditions to obtain the Blue Flag and Premier Seaside and Seaside resort awards (FEEE, 1995; TBG, 1995).

(Values in %)	European Blue Flag	TBG Seaside Premier Award	TBG Seaside Award
Clean beach	34.3	16.7	14.0
Clean water quality	34.5	14.4	11.1
Safety	25.9	16.0	12.4
Toilets	16.4	16.4	12.3
Popular	10.0	12.7	10.5
Sandy	12.7	14.1	10.7
Boating	8.9	9.6	7.8

Table 6.d.28 Perception of Beach Award Criteria

6.d.4.1 Summary Beach Flag Awards Results

A very high degree of confusion and lack of understanding existed with respect to beach award schemes at Whitmore Bay (Nelson *et al.,* in press (b)). No consistency of reply was evident. For example only 49% of respondents claimed to be aware of the different beach award systems, and yet 64% claimed that they were an important influence in their selection of a beach. This was further confounded by 60% of sample stating that they didn't know whether Whitmore Bay had attained one of these flags. It can be concluded that the Blue Flag has a higher profile amongst the public than the TBG Seaside Awards. For a discussion on Seaside Award Schemes see Section 6.e.7.

Chapter 6(e) Results and Discussion

6.e QUESTIONNAIRE SURVEY 1996

6.e.1 Part 1 Survey Response

The 1996 survey investigated three identified beaches along the South Wales coast, which differed in physical characteristics, with Cefn Sidan having superior water quality to the other two beaches (refer Chapter 2). Both beaches, Langland Bay and Cefn Sidan are of rural nature compared to Whitmore Bay cited on the resort of Barry Island. The survey work, totalling nine days, was carried out during warm to hot weather in August 1996 and a total of 821 responses were obtained (Tables 6.e.1 and 6.e.2). With respect to the statistical analysis, the Kruskal-Wallis Analysis of Variance on Ranks (Analysis of Variance) has been employed when the data sets failed the Kolmogorov-Smirnov Normality Test (P=<0.0001). Jandel Scientific (1995) and SPSS (1995) statistical packages have been used to run the tests.

Date	Location	H.Tide (Time)	Wind	Max. Temp. °C	Rainfall (mm)
6 Aug	Barry Island Cefn Sidan	12:34	Very Windy	17	25
7 Aug	Barry Island	13:15	Very Windy	16	0
8 Aug	Cefn Sidan	14:10	Very Windy	16	0
9 Aug	Barry Island	15:00	Windy	17	0
12 Aug	Cefn Sidan	18:16	Windy	20	0
13 Aug	Langland	19:04	Calm	23	0
14 Aug	Langland	18:52	Calm	23	0
17 Aug	Cefn Sidan	19:36	Calm	26	0
18 Aug	Barry Island	20:02	Calm	28	0

Table 6.e.1 Environmental Conditions During Survey 1996

Date	Location	Gender	Response	Accumulative Total
6 Aug	Cefn Sidan	Female	36	
	Cefn Sidan	Male	18	64
7 Aug	Whitmore	Female	103	
	Whitmore	Male	47	214
8 Aug	Cefn Sidan	Female	23	
	Cefn Sidan	Male	16	253
9 Aug	Whitmore	Female	81	
	Whitmore	Male	26	360
12 Aug	Cefn Sidan	Female	60	
	Cefn Sidan	Male	35	455
13 Aug	Langland	Female	106	
	Langland	Male	69	630
14 Aug	Langland	Female	48	
	Langland	Male	32	710
17 Aug	Cefn Sidan	Female	21	
	Cefn Sidan	Male	40	771
18 Aug	Whitmore	Female	37	
	Whitmore	Male	13	821

Table 6.e.2 Response Rate per Day for Beach Surveys

6.e.2 Demographic Characteristics for Total Survey

6.e.2.1 Age distribution

Similar to the 1995 survey, children under the age of 10 were excluded from the investigation as they were believed to be too young to provide responsible answers. Between the ages of 10-49 substantial numbers were obtained. Fewer numbers were achieved for the age range 50-59 and over 60s, but both were sufficient to facilitate statistical testing (Table 6.e.3). Analysis of Variance showed differences in the median values of the sample populations for age at the three beaches to be greater than would be expected by chance, indicating they are statistically significant different at the P = 0.001 level.

Stratifying the results by beach (Table 6.e.4), Whitmore Bay contained higher numbers of respondents between 20-49 with the highest density age range being 40-49 (23%). Similarly high numbers were recorded between the mid age groups for the 1995 survey at Whitmore Bay, made up mostly of parents. The highest percentages of young people were found at Langland Bay, with just under 50%of the sample being under 30 (49%) and just under 75% being under 40 (72%). Langland Bay is relatively accessible by public transport from Swansea and provides amenities for young people including tennis and also has good conditions for surfing, which may provide some of the reasoning for the large number of young people on the beach. The age distribution across all groups was fairly well balanced at Cefn Sidan, except for a low number over 60 years of age (4.6%). The Park and beach are geared for families, offering amenities and natural scenery for all ages which might account for the diverse range present at the beach.

Age Category (yrs)	Frequency	Percentage %
10-19	139	16.9
20-29	180	21.9
30-39	182	22.2
40-49	173	21.1
50-59	93	11.3
>60	54	6.6
Total (n)	821	100.0

Table 6.e.3 Distribution of Respondents Over Age 10

Age		10-19	20-29	30-39	40-49	50-59	>60	Total
Whitmore	Value	28	67	66	70	55	21	307
Bay	%	9.1	21.8	21.5	22.8	17.9	6.8	100.0
Langland	Value	62	64	58	42	8	21	255
Bay	%	24.3	25.1	22.7	16.5	3.1	8.2	100.0
Cefn	Value	49	49	58	61	30	12	259
Sidan	%	18.9	18.9	22.4	23.6	11.6	4.6	100.0

Table 6.e.4 Distribution of Respondents Age Stratified per Beach

6.e.2.2 Gender distribution

The total sample provided approximately 33.3% males and 66.6% females (Table 6.e.5), which is better balanced than the 1995 survey, which only yielded 22% of males. Field work observation indicated females to be more willing to participate in surveys. Analysis of Variance showed differences in the median values of the sample populations for gender at the three beaches to be greater than would be expected by chance, indicating they are statistically significant different at the P = 0.001 level. On stratification Whitmore Bay had a similar response rate for males as the 1995 survey with only 28%. Langland Bay was split 60:40 in favour of females and Cefn Sidan was well balanced with 46% male and 54% female (Table 6.e.6).

Gender	Frequency	Percentage %
Male	307	37.4
Female	514	62.6
Total (n)	821	100.0

Table 6.e.5 Distribution of Gender

Gender		Male	Female	Total
Whitmore Bay	Value	87	220	307
	%	28.3	71.7	100.0
Langland Bay	Value	101	154	255
	%	39.6	60.4	100.0
Cefn Sidan	Value	119	140	259
	%	45.9	54.1	100.0

Table 6.e.6 Distribution of Respondents Gender Stratified per Beach

6.e.2.3 Socio-Economic Status

Table 6.e.7 details the breakdown of the whole sample in terms of their socio-economic status. The employed category ranked 1 with 47%, nearly 50% of the total sample. Housewives and students were also well supported with 19% and 24% respectively. Low numbers were recorded for the unemployed and retired category. Analysis of Variance showed differences in the median values of the sample populations for socio-economic status at the three beaches to be greater than would be expected by chance, indicating they are statistically significant different at the P = 0.05 level. Table 6.e.8 shows there was not a great deal of difference in the proportion for the employed, unemployed and retired categories at all three beaches. The major differences occurred for the house wife group with Whitmore Bay having the highest number of respondents (29.3%) and students with Langland Bay having the highest number of respondents (32.5%), in line with the large number of young people at the beach.

Socio-economic Status	Frequency	Percentage %
Employed	383	46.7
House Wife	153	18.6
Student	198	24.1
Unemployed	27	3.3
Retired	60	7.3
Total (n)	821	100.0

Table 6.e.7 Distribution of Socio-Economic Status

Age		Employed	H.Wife	Student	Unemployed	Retired	Total
Whitmore	Value	133	90	49	12	23	307
Bay	%	43.3	29.3	16.0	3.9	7.5	100.0
Langland	Value	113	27	83	9	23	255
Bay	%	44.3	10.6	32.5	3.5	9.0	100.0
Cefn	Value	137	36	66	6	14	259
Sidan	%	52.9	13.9	25.5	2.3	5.4	100.0

Table 6.e.8 Distribution of Respondents SES Stratified per Beach

6.e.2.4 Geographical Distribution of Respondents

Table 6.e.9 shows the distribution of visitor type to be well matched with just over 33% of respondents being attributed to the holiday group (35%) and day tripper (36%). The lowest category were locals with 29%. Analysis of Variance showed differences in the median values of the sample populations for visitor type at the three beaches to be greater than would be expected by chance, indicating the differences to be statistically significant at the P = 0.001 level (Table 6.e.10). Whitmore Bay is a resort beach with high degree of facilities including a holiday camp accounting for the large number of holiday makers (35%) and day trippers, which almost made up 50% of the sample (47%). The local category was low (18%). However, Barry has five other beaches, which tend to be heavily frequented by locals. The 1995 survey at Whitmore Bay also showed a significant presence of non-locals. Langland Bay had equal ratios of holiday makers and day trippers (23%) with locals making up 54%. Cefn Sidan had even less locals than Whitmore Bay (15%) with the largest number of respondents on holiday (48%) with day trippers making up the other 37%.

Origin	Frequency	Percentage %
Holiday	288	35.1
Day Tripper	299	36.4
Live locally	234	28.5
Total (n)	821	100.0

Table 6.e.9 Distribution of Visitor Type

Visitor Type		Holiday	Day Visitor	Local	Total
Whitmore Bay	Value	107	145	55	307
	%	34.9	47.2	17.9	100.0
Langland Bay	Value	58	58	139	255
	%	22.7	22.7	54.5	100.0
Cefn Sidan	Value	123	96	40	259
	%	47.5	37.1	15.4	100.0

Table 6.e.10 Distribution of Respondents Visitor Type Stratified per Beach

6.e.3 Perception of Female Hygiene Items and Condoms

6.e.3.1 Recognition of Female Sanitary Item

Using a photograph of a sanitary towel the respondents were asked to write down what they believed the item to be. When cross tabulated with gender it is apparent that females have a significantly higher recognition than males (Table 6.e.11), verified using Chi-square at the P = 0.05 level. Over 88% of females recognised the photograph as a sanitary towel, with 6% believing it to be a plaster. Conversely, 71% of males accurately perceived the photograph as being a sanitary towel, with 26% believing it to be a plaster. Analysis of Variance showed no statistical difference across the three beaches (P=>0.05).

When managing perception to litter and sewage-related debris it is essential to be aware of the recognition and impact on the beach user.

Gender	Sanitary Towel	Plaster	Paper	Condom	Row Total
Male	218	79	4	6	307 37.4
Female	453	50	5	6	514 62.6
Column Total	671 81.7	129 15.7	9 1.1	12 1.5	821 100.0

Table 6.e.11 Chi-square - Recognition of a Sanitary Item vs. Gender

Chi-Square	Value	DF	Significance
χ^2	39.23604	3	0.000

6.e.3.2 Condom Equivalent

To compare the difference in visual impact of litter items found on a beach, the respondents were shown photographic plates of a range of litter items and asked to rate on a scale of 1-9 how offensive each plate was. One indicated not very offensive, 9 indicated very offensive (Appendix II). Condoms were used as reference. The mean values of the other litter items were divided by the mean value calculated for condoms, to give a comparison ratio. This formed what was called the condom equivalent (CE). Table 6.e.12 details the CE for each item for the total sample. Sanitary towels were perceived to be almost as offensive as condoms (CE = 0.99), both being sewage-related debris. Aluminium cans scored second highest with a CE = 0.71 followed by a tie of 0.66 for both plastic bottles and polystyrene containers. Crisp packets were the least offensive item with a CE of 0.62. Table 6.e.13 details the computation of all CE values for each beach. Analysis of Variance showed differences in the median values of the sample populations for the different beaches not to be greater than would be expected from random sampling (P = >0.05). This concept has not been investigated before, and has potential use for grading beach pollution. Further work is required to support the use of this technique. Even though all three beaches showed little variation, it would be prudent to investigate a greater range of items for a larger selection of beaches.

Column	Size	CE	Mean	Std Dev	K-S Distance	P Value
Sanitary Towel	821	0.99	8.516	1.227	0.450	<0.001
Ally Can	821	0.71	6.129	2.242	0.132	<0.001
Plastic Bottle	821	0.66	5.682	2.251	0.0982	<0.001
Condom	821	1.0	8.608	1.219	0.468	<0.001
Crisp Packet	821	0.62	5.302	2.371	0.0941	<0.001
Polystyrene	821	0.66	5.705	2.353	0.112	<0.001

Table 6.e.12 Condom Equivalent for the Total Sample

Column	Whitmore Bay	Langland Bay	Cefn Sidan	Total Sample
Sanitary Towel	0.99	1.00	0.97	0.99
Ally Can	0.73	0.71	0.69	0.71
Plastic Bottle	0.66	0.67	0.66	0.66
Condom	1.00	1.00	1.00	1.0
Crisp Packet	0.61	0.64	0.60	0.62
Polystyrene	0.64	0.69	0.66	0.66

Table 6.e.13 Comparison of Condom Equivalent Values at the Three Beaches

6.e.4 Part 2 Attitudes to Beach Awards

The 1996 survey further developed the work on beach awards that was conducted in 1995 (Section 6.d.4) at an additional two beaches, Langland Bay and Cefn Sidan. Analysis of the 1996 survey compared the recognition, knowledge and understanding of seaside award schemes, including the MCS Good Beach Guide across three beaches in South Wales. The three beaches have very different physical characteristics. Whitmore Bay is a large resort beach, with poor water quality, which receives high volumes of beach users during the summer months (refer Section 2.2). Langland Bay is also very popular in the summer months, but has less facilities than Whitmore Bay. The water is generally of good quality (see Section 2.3.1). Cefn Sidan has very few facilities, although it is situated in a well managed Country Park, which offers certain services. The water quality is excellent and Cefn Sidan was the only one of the three beaches to receive a seaside award, attaining both the Blue Flag and Seaside Award Status during 1996 (see Section 2.4).

The TBG changed their beach flags system for the 1996 season (TBG, 1996). The Premier Award was dropped because it was believed to be superfluous with respect to holding virtually the same criteria as the Blue Flag, only serving to complicate and confuse the public (Hines pers.comm., 1995; Nelson *et al.,* 1997a, 1997b). The Seaside Award was retained for both resort and rural beaches, catering for beaches that failed the EC Guideline standards, but met the Mandatory standards. In addition questions relating to the MCS Good Beach Guide were introduced into the survey, not included in the 1995 work (MCS, 1996). Questions relating to beach award systems were ordered in a sequence to establish an accurate level of understanding of the knowledge and recognition respondents had towards these schemes at the three beaches, and attempt to ascertain their worth as marketing tools. As previously stated in section 6.d.4 there is a paucity of literature relating to the perception of beach award schemes.

6.e.4.1 Awareness of Beach Rating Schemes

The respondents were asked if they were aware of the various beach rating schemes. From the total sample 53% claimed to know something about the different beach award systems. When broken down per beach, there was a very low acknowledgement at Whitmore Bay (42%), which was down 5% from the 1995 survey. The highest response was at Cefn Sidan (62%), the only beach to fly both the TBG Seaside Award and Blue Flag. Cefn Sidan also had the highest volume of holiday makers present at the beach. Obviously these had decided to choose the location on preference rather than convenience, as opposed to day visitors and locals. It could be argued that those on holiday had selected Cefn Sidan due to their knowledge of the consistently good beach quality there and recognition of the flags achieved.

The main two seaside flag awards operating in the UK are the TBG Seaside Award and the EC Blue Flag. It is evident from Table 6.e.14 that over one and a half times as many respondents (1.65) from the total sample had heard of the Blue Flag compared to the Seaside Award, verified statistically using Analysis of Variance ($P = <0.001$). The 1995 survey at Whitmore Bay also revealed more awareness amongst respondents of the Blue Flag than the Seaside Award. Fifty five percent of the total sample claimed to be aware of the MCS Good Beach Guide.

Table 6.e.15 highlights that the awareness of the Blue Flag and Seaside Award vary significantly across the three beaches, verified statistically using the Analysis of Variance ($P = <0.001$). However, there was not a statistically significant variation in awareness of the Good Beach Guide across the three beaches (Analysis of Variance, $P = 0.058$) This may be attributed to the fact this system does not include visual display of a beach flag. In general there was very low awareness of the Seaside Award at all three beaches, falling below 40% at Whitmore Bay and Langland Bay. Whitmore Bay scored the lowest for both Good Beach Guide (50%) and Blue Flag (53%) and Langland Bay recording the lowest score for the Seaside Award (32%). The highest response for the Good Beach Guide and Seaside Award were obtained at Cefn Sidan, with respective socres of 60%

and 52%. The highest score for the Blue Flag was achieved at Langland Bay with a response of 75%.

Values in %	Yes	No	Unsure	Missing Value
Good Beach Guide	55	28	13	4
Blue Flag	66	23	8	3
Seaside Award	40	38	17	5

Table 6.e.14 Recognition of Beach Rating Schemes, Total Sample

Values in %	Good Beach Guide	Blue Flag	Seaside Award
Whitmore Bay	50	53	37
Langland Bay	57	75	32
Cefn Sidan	60	70	52

Table 6.e.15 Recognition of Beach Rating Schemes at the Three Beaches

6.e.4.2 Influence of a Beach Award Status on Beach Choice

The respondents were asked if either The Good Beach Guide, Blue Flag or Seaside Award schemes had influenced their choice of beach. Even though in most cases some 50% of the total sample had heard of these systems (Table 6.e.14), approximately 40% claimed to be influenced in beach selection to them (Table 6.e.16). The data values displayed in Table 6.e.16 are almost identical, showing no statistical difference, confirmed by using Analysis of Variance ($P = 0.489$). Table 6.e.17 shows the influence of the schemes, stratified per beach. Very little consistency of pattern emerged from the data. Cefn Sidan scored highest in all three categories with the Good Beach Guide scoring 50%, followed by the Blue Flag, 48%. The highest score at Whitmore Bay was for the Good Beach Guide (40%) and for Langland Bay, the Blue Flag (38%). It must be noted that the question did not relate specifically to the actual beach the interview took place. However, there does appear to be a link between the recognition and importance of beach systems at Cefn Sidan, which has ranked top for this and the preceding questions.

Values in %	Yes	No	Unsure	Missing Value
Good Beach Guide	40	33	19	8
Blue Flag	40	35	19	6
Seaside Award	37	35	20	8

Table 6.e.16 Influence of Beach Rating Schemes on Beach Selection, Total Sample

Values in %	Good Beach Guide	Blue Flag	Seaside Award
Whitmore Bay	40	33	37
Langland Bay	31	38	29
Cefn Sidan	50	48	46

Table 6.e.17 Influence of Beach Rating Schemes at the Three Beaches

6.e.4.3 Effect of Beach Attributes on Beach Selection

Respondents were given a list of attributes (Table 6.e.18) related to a beach and asked to place them in rank order, with one being the most important and six the least important. Similar attributes were listed in the 1995 survey at Whitmore Bay, but respondents were asked to highlight the three most important in beach choice, in contrast to using rank order. The style of question was altered for the 1996 survey to the use of ranking as it lends itself more to statistical analysis than the method used in the 1995 survey. Analysis of Variance was employed which showed differences in the median values among the treatment groups to be greater than would be expected by chance, indicating a statistical difference between the attributes ($P = <0.001$).

Some of these attributes are necessary requirements for the above awards, notably good water quality (Appendix VI). Table 6.e.18 highlights the top two ranks for each attribute. Only 15% placed flag as ranking first in importance. Water quality was the highest ranking attribute with (41%) followed closely by clean sand (37%). Facilities, distance and views and landscape all have low scores with 6%, 6% and 5% respectively. Table 6.e.19 compares the top rank score for each attribute stratified across the three beaches. The data values describing the beach attributes displayed in Table 6.e.19 show very little variation, confirmed statistically by using Analysis of Variance on Ranks ($P = >0.05$). The only exception was distance travelled which proved to be statistically different ($P = <0.001$). This is can probably be ascribed to the significantly higher proportion of locals at Langland Bay compared to the other two beaches. The highest scores were recorded for water quality and clean sand for all beaches. Due to the style of question being different from the 1995 survey, it is not possible to directly compare results. However, it is worth noting that the results obtained at Whitmore Bay, 1995, were vastly different with distance recording the highest score (76%) and clean water scoring a low (7%). It is not immediately apparent why these two sets of results are so disparate.

Values in % (n=821)	Rank 1	Rank 2
Beach Award Flag	15	6
Clean Water	41	37
Clean Sand	34	38
Facilities	6	10
Distance Travelled	6	4
Views and Landscape	5	4

Table 6.e.18 Importance of Beach Attributes, Rank 1 and 2

Values in %	Whitmore Bay	Langland Bay	Cefn Sidan
Beach Award Flag	14	15	16
Clean Water	39	42	43
Clean Sand	33	31	37
Facilities	6	4	6
Distance Travelled	4	10	4
Views and Landscape	3	9	5

Table 6.e.19 Importance of Beach Attributes, Comparison of Beaches (Rank 1)

6.e.4.4 Visual Identification of Beach Award Systems

The Blue Flag and Seaside Award flags are flown at beaches which have applied for and successfully attained one of these awards. To establish the level of recognition beach users have with the Blue Flag and Seaside Award flag the respondents were shown them with a selection of other flags (Table 6.e.20). They were asked to match each flag with its corresponding meaning. Table 6.e.20 shows that a very low proportion of the total sample correctly identified the Blue Flag (26%) and Seaside Award flag (29%). The most accurate responses were for the Swedish flag (66%) and lifeguard flag (57%). Also under half of the sample accurately recognised the European Union Flag (49%). As mentioned earlier, a mini study by Paul (*pers.comm.*, 1997) at Lime Regis showed over 72% failed to recognise either the Blue Flag of Seaside Award.

The data response for each beach regarding visual identification of the flags, Table 6.e.21, all followed normal distributions, verified using the Kolmogorov-Smirnov Test (Whitmore Bay P=0.708; Langland Bay P=0.378; Cefn Sidan P=0.4052). One Way Analysis of Variance was applied to the data which proved there were no statistically significant differences in response to accurate identification of the flags between Langland Bay and Cefn Sidan, but both differed from Whitmore Bay at the P = 0.05 level. The only exception was the European Union Flag where no statistical significant difference was observed across the three beaches, that were great enough not to exclude the possibility that the difference is due to random sampling variability (P = 0.803).

Values in %	Correct	Unsure
Blue Flag	25.6	20.1
Swedish Flag	66.0	20.2
European Union Flag	48.5	21.2
Seaside Award Flag	28.9	22.0
Lifeguard Patrol Flag	57.1	24.0

Table 6.e.20 Visual Flag Identification for Total Sample

Values in %	Whitmore Bay	Langland Bay	Cefn Sidan
Blue Flag	18.9	29.4	29.7
Swedish Flag	53.7	72.9	73.7
European Union Flag	35.8	56.1	56.0
Seaside Award Flag	24.8	31.0	31.7
Lifeguard Patrol Flag	41.4	65.5	67.6

Table 6.e.21 Comparison of Visual Flag Identification at the Three Beaches

6.e.4.5 Identification of Beach Award Scheme Criteria

The participants in the study were asked to identify a set of criteria with the corresponding seaside award scheme, to establish the level of understanding for each. The listed criteria were not a comprehensive set of attributes for all the schemes, but a select few similar to those used in the 1995 survey were used (refer Section 6.d.4, Table 6.d.28). Two dummy variables, sandy beach and boating facilities were added which were not related to any of the awards. In contrast to the original survey the TBG Premier Seaside Award, no longer in existence, was replaced by the Good Beach Guide. The Good Beach Guide concentrates mainly on water quality but also lists other attributes of beaches, such as facilities and toilets. A full set of attributes for these systems are detailed in Appendix VI.

Table 6.e.22 gives the percentage response for each of the awards. A statistical difference was observed between the responses obtained from all three schemes, beyond what might be expected from random sampling variability, supported using Analysis of Variance on Ranks ($P = <0.001$). Analysis of Variance on Ranks was also used to investigate statistical differences between the responses for the three schemes across the three beaches. It was proved that there was not a statistical variation beyond what might be expected from Random Sampling at the $P = >0.05$ level.

There was a higher tendency for participants to identify the Blue Flag with clean water (51%) and clean beach (57%) than both the Seaside Award and Good Beach Guide. In a study by House and Herring (1995) 41% correctly identified the Blue Flag with clean water and 27% though the Seaside Award meant the water met EC Mandatory levels. Although the response rate was generally under 50% for all eight attributes, the dummy variables for the Blue Flag scored very low, boating 8% and sandy beach 14%. Few respondents related the Blue Flag with being a tourist beach (16%), which is an essential attribute, whereas a higher proportion identified the Seaside Award (28%) and Good Beach Guide (38%) with being tourist beaches, which is not necessarily true.

Data obtained at Whitmore Bay for the Blue Flag for seasons 1995 and 1996 were tested against each other to investigate if a statistical difference existed (Table 6.e.23). The Mann Whitney Rank Sum Test showed the difference to be negligible at the ($P=0.29$) beyond what might be expected from Random Sampling. The Mann Whitney Rank Sum Test was performed on the Seaside Award data, which showed there to be a statistical difference ($P = <0.001$). However this should be treated with caution as the Seaside Award schemes changed over the 1995-1996 season. Comparisons for the Good Beach

Guide could not be carried out, as no data was collected for the 1995 season on the scheme.

(Values in %)	European Blue Flag	TBG Seaside Award	Good Beach Guide
Clean Beach	51.2	40.8	43.7
Clean Water Quality	56.7	34.8	30.6
Safety	43.8	37.7	41.5
Toilets	19.6	37.4	34.9
Tourist Beach	15.9	28.2	38.2
Sandy	14.2	32.7	34.7
Boating	8.3	15.8	25.6
Dogs Banned	31.3	25.9	34.5

Table 6.e.22 Perception of Beach Award Criteria

(Values in %)	European Blue Flag 1996	European Blue Flag 1995	TBG Seaside Award 1996	TBG Seaside Award 1995
Clean beach	43.6	34.3	36.2	14.0
Clean water quality	45.3	34.5	31.9	11.1
Safety	40.4	25.9	33.2	12.4
Toilets	20.5	16.4	34.6	12.3
Popular	19.3	10.0	25.7	10.5
Sandy	16.7	12.7	29.7	10.7
Boating	10.4	8.9	15.3	7.8

Table 6.e.23 Comparison of Data for Flag Schemes Between 1995 and 1996, Whitmore Bay

6.e.4.6 Summary of Attitudes to Beach Rating Awards Results

Results obtained regarding knowledge and understanding of beach award systems at Whitmore Bay, Langland Bay and Cefn Sidan confirmed findings from the 1995 survey, which suggested a general confusion over their meaning. The Blue Flag proved to be more widely heard of at all three beaches in comparison to the Good Beach Guide, which in turn received more recognition than the Seaside Award. These findings were in agreement with the 1995 survey, which also found highest recognition attributable to the Blue Flag compared to the Seaside Award. However, contradictory findings emerged over the high number of respondents (*circa* 40%) who claimed that one of these systems influenced their selection of beach, but when asked what attributes of a beach were important approximately only 15% responded by stating a beach award.

Further inconsistencies emerged when the respondents were asked to identify the Blue Flag and Seaside Award with the appropriate flag, shown on photographic plates. Only 26% recognised the Blue Flag and 29% recognised the Seaside Award. In view of the higher number of people who had greater knowledge of the Blue Flag than the Seaside Award Flag, it was surprising that more people accurately noted the Seaside Award flag. This may well be due to the construction of the Seaside Award flag being more representative of the beach, showing sand and sea, with the Blue Flag being only dual coloured, blue and white.

When asked to highlight on a list attributes applicable to the different beach systems, clean water and beach scored highest with a higher ratio of participants responding to the Blue Flag. With regard to recognition at different beaches of the systems, there was definitely a greater response at Cefn Sidan, the only beach to have successfully achieved the Blue Flag, Seaside Award flag and be mentioned in the Good Beach Guide. The lowest recognition of the systems was at Whitmore Bay, However, even at Cefn Sidan only 30% recognised the Blue Flag and 32% recognised the Seaside Award. Although 37% claimed flags to influence their choice of beach, only 16% noted flags to be important in comparison with other beach attributes.

6.e.5 Part 3 Attitudes to Water Quality

6.e.5.1 Turbidity

The Secchi disc is used to measure the transparency of water which is a function of turbidity, which impairs clarity (Internet, 1997a), refer to Chapter 5, Section 5.1.3. Suspended solids in the water, including silt, sewage, plankton and industrial wastes reduce the transmission of light (Internet, 1997b). It is believed that the clearer the water the more desirable for swimming (National Academy of Sciences, 1973). Secchi disc readings were taken on high tides during calm, flat tidal conditions over a period of days and at various points across the beach to compare the relative turbidity at Whitmore Bay, Langland Bay and Cefn Sidan. The mean values are given in Table 6.e.24 along with their standard deviation and maximum and minimum values. Langland Bay had a visibility 1.6 times greater than Whitmore Bay and Cefn Sidan had a visibility of 2.0 times greater than Whitmore Bay.

No literature was found relating perception of turbidity in coastal waters to aesthetic quality or pollution, although work has been done on inland waterways, particularly lakes (Anon., 1987; Francis *et al.,* 1994; Smith *et al.,* 1995a, 1995b; Carlson, 1995; Phillip, 1996). Smith *et al.,* (1995b) did research on lakes in New Zealand and found water to be suitable for bathing at Secchi disc readings of up and above 1.5m. In Canada primary contact recreation waters must reach a Secchi disc depth (SD) of 1.2m (cited Phillip, 1996), slightly less than that quoted by Smith *et al.,* (1995b). The EC Bathing Water Directive stipulates a Mandatory SD standard of 1m and Guideline SD standard of 2m (CEC, 1976a). Both Langland and Cefn Sidan would reach the EC Mandatory standard and Langland Bay would just fail the requirements quoted by Smith *et al.,* (1995b). However, Whitmore Bay would fail all standards.

Beach	Mean (m)	Std Dev	Max	Min
Whitmore Bay	0.92	0.06	0.98	0.87
Langland Bay	1.45	0.07	1.53	1.39
Cefn Sidan	1.85	0.17	2.04	1.72

Table 6.e.24 Secchi Disc Readings for the Three Beaches

6.e.5.2 Perception of Water Quality and Turbidity

The participants were asked whether they agreed or disagreed with the statement that 'murky water indicates poor water quality', on a scale of 1-5 (Table 6.e.25). One indicated strongly agree and five indicated strongly disagree. By combining categories one and two the pooled data suggested that the opinion of beach users was weighted towards agreement that murky water is perceived to related to poor water quality. At Whitmore categories one and two showed 56% of respondents to agree with the statement compared to 22% who disagreed, 53% agreed at Langland compared to 30% who disagreed and 50% agreed at Cefn Sidan compared to 31% who disagreed. The Kruskal-Wallis One Way Analysis of Variance on Ranks was applied to the response data, proving that no statistically significant difference existed between the perception of murky water across the three beaches, outside what could be attributed to random sampling variability ($P = 0.237$). The data sets for the three beaches all failed the Kolmogorov-Smirnov normality test ($P = <0.001$). These findings were in agreement with David (1971) and Nicolson and Mace (1974) who both reported 26% and 35% respectively, of their samples to claim murky water to be related to poor water quality. Later work by Dinius (1981) and House and Sangster (1991) further supported the idea that clarity is perceived to be an indicator or water pollution.

Values in %	Whitmore Bay	Langland Bay	Cefn Sidan	Total Sample
1. Strongly Agree	24.1	20.4	17.4	20.6
2. Agree	31.6	32.9	32.4	32.3
3. Don't Know	17.6	14.9	18.1	16.9
4. Disagree	19.5	25.1	26.6	23.7
5. Strongly Disagree	2.3	4.7	4.2	3.7
Missing Value	4.9	2.0	1.2	2.7

Table 6.e.25 Comparison on Perception of Murky Water Between the Three Beaches

Respondents were asked to rate the clarity of the water at the respective beaches, Whitmore Bay, Langland Bay and Cefn Sidan on a scale of 1 to 9. One being very clear and 10 indicating very murky. Table 6.e.26 shows the mean and standard deviation values for the three beaches. It is clear that beach users at Whitmore Bay considered the water to be more turbid than that at both Langland Bay and Cefn Sidan with a mean of 6.5. The mean value at Langland Bay was the lowest with 4.2 and Cefn Sidan with 4.9. The Kruskal-Wallis One Way Analysis of Variance on Ranks supports these variations statistically by indicating that the distributions of the three beaches vary significantly, verified at the $P = <0.001$ level. The data sets all failed the Kolmogorov-Smirnov Normality test ($P = <0.001$). The results of this question show beach users are sensitive to water clarity, with Whitmore Bay being considerably more turbid than the other two beaches (Table 6.e.24).

Column	Size	Mean	Std Dev
Whitmore Bay	307	6.511	2.183
Langland Bay	255	4.227	1.749
Cefn Sidan	259	4.907	2.158

Table 6.e.26 Comparison on Perception of Water Clarity at the Three Beaches

Respondents were also asked to rate the water quality at the respective beaches on a scale of 1 to 9. One being very clean and 9 indicating very dirty. Table 6.e.27 shows the mean and standard deviation values for the three beaches. The beach users at Whitmore Bay perceived the water to be considerably more dirty (mean 6.1) than at Langland Bay (mean 4.2) and Cefn Sidan (mean 4.9). The Kruskal-Wallis One Way Analysis of Variance proved this difference to be statistical significant at the $P = <0.001$ level. No statistical difference was observed between the distributions of data at Langland Bay and Cefn Sidan ($P = >0.05$). The data sets all failed the Kolmogorov-Smirnov Normality test ($P = <0.001$). The findings of these results show beach users to perceive the water quality at Whitmore Bay to be considerably lower than the other two beaches, which might be due to the lack of clarity at Whitmore Bay. Research has shown a positive association between of lack of water clarity and discoloured water and perception of poor water quality (Herzgog, 1985; Burrows and House, 1989; House and Sangster, 1991; Green and Birchmore, 1993).

Column	n	Mean	Std Dev
Whitmore Bay	307	6.052	2.114
Langland Bay	255	4.647	1.891
Cefn Sidan	259	4.390	2.760

Table 6.e.27 Comparison on Perception of Water Quality at the Three Beaches

6.e.5.3 Beach User Behaviour

Beach users behaviour was investigated in relation to their affinity to water or willingness to participate in activities with increasing water contact at the three beaches. The questionnaires were mostly conducted on dry sand, and the categories were not mutually exclusive. Table 6.e.28 highlights the significant differences between percentage activity levels across the three beaches, verified statistically using the Kruskal-Wallis One Way Analysis of Variance on Ranks test. All test runs were significant at the $P=<0.05$ level, except for the swimming category which was significant at the $P=<0.01$ level. The data sets all failed the Kolmogorov-Smirnov Normality test (P = <0.001). Respondents at Langland were more likely to involve themselves with water activities at all levels in contrast to respondents at Whitmore Bay. For example, at Langland Bay 82% of beach users were prepared to walk on west sand and 28% were prepared to immerse their head whilst swimming, compared to Whitmore Bay, where only 66% were prepared to walk on wet sand and less than 8% were prepared to immerse their head whilst swimming. Data scores for Whitmore Bay were the lowest for every category, except foot paddling. Data scores for Cefn Sidan were almost mid-way between Whitmore and Langland Bay for most categories. The largest variance between scores for the three beaches occurred for wading, swimming and swimming with head immersion. For example there was nearly a twice fold percentage of respondents willing to swim at Langland Bay than at Whitmore Bay. Again these differences could well be explained by perception of beach users that turbid water indicates polluted water. Ditton and Goodale (1974) commented that naturally turbid waters are often perceived to be dirty, even when they are of good quality. Other contributory factors might be due to the higher percentage of younger people at Langland Bay, who are more likely to participate in water activities (refer Section 6.b.1). Although the turbidity readings at Langland Bay and Cefn Sidan were similar, beach users were still much less likely to involve themselves in water activities at Cefn Sidan compared to Langland Bay. One factor which might account in part for this is that at intertidal distance is much greater at Cefn Sidan, compared to Langland Bay (refer Chapter 2).

Values in %	Whitmore Bay	Langland Bay	Cefn Sidan
n	307	255	259
Walk on wet sand	66.1	82.4	81.1
Stand on waters edge	71.7	81.2	72.6
Foot paddle	66.1	77.6	63.7
Wade to knee depth	39.1	60.0	51.7
Swim not immerse head	17.6	31.4	26.3
Swim - immerse head	7.8	28.2	22.0

Table 6.e.28 Comparison of Beach Behaviour at the Three Beaches

6.e.5.4 Perception of Sea Pollution

Respondents were given a list of pollutants found in the sea (Table 6.e.29) and asked to place them in rank order, with one being the most offensive and five the least offensive. A similar question was included in the 1995 survey at Whitmore Bay, but respondents were asked to highlight the three most offensive sea pollution items from a list, in contrast to using rank order. The style of question was altered for the 1996 survey to the use of ranking as it lends itself more to statistical analysis than the method used in the 1995 survey. Table 6.e.29 highlights the top rank for each category for the total sample. Oil (56%) scored considerably higher than any other value in being perceived as the most offensive sea pollutant, which is in contrast to findings from the 1995 survey, which showed only 35% of respondents to quote oil as an offensive sea pollutant. Morgan and Williams (1995) and the Robens Institute (1987) also found oil to figure prominently in reports of coastal pollution. Floating debris (21%) consisting of anything from food items, faeces and sanitary towels to drift wood ranked second followed by foam/scum (18.4%) in third, mainly being surfactants. Research conducted by David (1971) also found floating objects to score highly (20%) in terms of visual pollution, backed by work done by Nicolson and Mace (1974). And the Robens Institute (1987) found a similar proportion of their sample (22%) to state foam/scum to be objectionable. More recent work by Young *et al.,* (1996) used a scale of 0-9 to rate the preferences and priorities of beach users to beaches. In congruence with these findings their results also showed concern over the presence of floating objects, in particular sewage-related debris and oil contamination. It was not possible to compare these results directly with the 1995 survey due to the difference in question style. However, the pollutants were listed in exactly the same order, except for oil which was perceived to be less offensive in 1995 survey.

Table 6.e.30 lists the top ranked category for Whitmore Bay, Langland Bay and Cefn Sidan. The Kruskal-Wallis One Way Analysis of Variance on Ranks was employed which showed the differences in the median values among the beaches were not statistically different for the categories discoloured water, foam/scum and floating debris at the $P = >0.05$ level. However, the analysis showed the values for smell and oil were found to be statistically different beyond what might be expected due to random sampling (P =

<0.05). In both cases higher scores were recorded at Cefn with smell (14.3%) and oil (62%) in contrast to Whitmore Bay which had the lowest scores for these two categories with smell (11%) and oil (54%). All three beaches failed the Kolmogorov-Smirnov Normality test (P = <0.001).

Values in % (n=821)	Rank 1
Oil	56
Foam/scum	41
Floating debris	34
Discoloured water	15
Unusual smell	6

Table 6.e.29 Perception of Sea Pollution, Rank 1

Values in %	Whitmore Bay	Langland Bay	Cefn Sidan
Discoloured water	9.8	7.8	5.8
Unusual smell	10.7	11.4	14.3
Foam/scum	18.2	23.1	13.9
Floating debris	20.8	22.2	19.7
Oil	53.7	54.1	61.8

Table 6.e.30 Perception of Sea Pollution (Rank 1), Comparison of Beaches

6.e.5.5 Summary of Turbidity, Behaviour and Attitudes to Water Quality Results

Secchi disc turbidity readings were taken at all three beaches, to gauge beach behaviour in relation to visual impairment of water quality (see Section 7.4). Mean turbidity readings were significantly higher at Whitmore Bay, 0.92m, compared to Langland Bay and Cefn Sidan, with readings of 1.45m and 1.85m respectively. The high turbidity in the

Bristol Channel is due to the high sediment load from the Severn Estuary (Severn Estuary Strategy, 1997). With increasing distance, moving west from the Severn Estuary, turbidity decreases. This would explain why the clarity was so low at Whitmore Bay, followed by Langland Bay and finally the highest Secchi disc readings attained at Cefn Sidan.

Results confirmed general concern over coastal pollution, emanating mostly from media coverage. Seventy percent perceived the sea water at Whitmore Bay to be dirty and 55% claimed that the water was too unclean to swim in. When asked to state the most offensive forms of sea pollution, floating objects were recorded with the highest score of 76%, from the total sample (N=1038), which included sewage-related debris such as condoms and sanitary towels and general litter items. Discoloured water also ranked highly with 58% of the sample claiming the water at Whitmore Bay to lack clarity. When asked to comment on the most observed debris items on beach, food packaging including plastic bottles, aluminium cans scored the highest on both the shore and in the sea. Sewage-related debris items were more prevalent in the sea than the beach. These results tie in with the litter recorded in the strandline and transect analysis. The second survey also showed floating debris to figure prominently being perceived to be visually very offensive along with oil and discoloured water. These findings are in agreement with work by the Robens Institute (1987), Nicolson and Mace (1975) and David (1971) who all found floating objects and discoloured water as being very offensive to beach users.

Over 50% of the sample believed murky water indicated poor water quality, with the highest score attained at Whitmore Bay, which also had the highest turbidity readings. When asked to rate the clarity of the water at the respective beaches, beach users were reasonably accurate in their assessment. Again Whitmore Bay had a significantly higher response rate, indicating poor clarity, followed by Langland Bay and the lowest score was achieved by Cefn Sidan, having the lowest turbidity readings. Results also suggested that with increasing turbidity there is an increasing reluctance to make contact with sea water. Whitmore Bay proved consistent with attaining the largest number of respondents unwilling to enter into water based activities, only 18% stated they would be prepared to swim.

6.e.6 Discussion of Results on Perception to Coastal Pollution, Beach User Behaviour Analysis and Aesthetic Indicators for Surveys 1995 and 1996

Definitions of aesthetics and perception are inherently related as defined by the Oxford Dictionary (1991). Aesthetic quality relates to that which can be detected by the senses, including visual, audible, olfaction, touch and taste. Of these visual appearance is the most important to control, which was highlighted by Everard (1995) who also stated that visual appearance is the most significant factor in public perception of water quality, followed by odour of water bodies. The WHO (1994a) and Williams and Nelson (1997) identified the need to set aesthetic quality indicators to protect the psychological well-being of the beach user in addition to conventional physio-chemical and microbiological determinands set to safeguard physical health. Failure to control the aesthetic quality of the coastline, apart from affecting tourism revenue, seriously affects the experience of a visit to the beach.

Results of the grid analysis (refer Section 6.c.1) found a mixture of litter including general visitor discards and sewage-related debris to be more visually offensive than the separate generic categories of debris (Williams and Nelson, 1997). This finding is at variance with other research, including Green and Birchmore (1993) who noted sewage-related debris to be the most offensive coastal debris. However, the condom equivalent analysis showed that condoms, followed closely by sanitary towels are independently the most offensive items found on a beach. Females were found to be significantly more sensitive to coastal pollution than males. Females also showed approximately 25% higher recognition of sanitary hygiene items than males.

Management of litter on the beach has been discussed and regular cleansing is required to ensure a clean beach. Further work on aesthetic indicators will help in formulating grading frameworks to assess beach quality in terms of debris. Development of the NALG ABCD Model (Earll and Jowett, 1998) is a positive advancement in the right direction (see Section 4.a.6.1). The use of photographic plates to measure perception of aesthetic quality was piloted in this study. This technique is being widely implemented and proving to be an effective instrument (Dinius, 1981; Hertzgog, 1985; Williams and

Lavelle, 1990; House and Herring, 1995), but needs further development. For rural beaches which do not have mechanical cleansing, stewardship schemes should be encouraged. Control of marine debris originating from aquatic sources is different issue and very difficult to manage. In particular high natural turbidity, which does not necessarily indicate poor water quality, has a negative impact on perception of water quality and effects beach behaviour. This view was supported by the WHO (1994a) who stated that poor aesthetics indicates polluted water quality and has led in some cases to an increase in reported gastrointestinal complaints. Ditton and Goodale (1974) and Smith *et al.* (1995b) found visual quality of the water to affect judgement of water quality, with discoloured water being perceived as dirty.

The only real solution is to attempt to educate the public and ensure water quality is good (Williams and Nelson, 1997). Grant and Jickells (1995) share this view point, stating education is a necessary component to achieve a cleaner marine environment, but also highlighting the virtually impossible task of policing seafaring craft. Sewage discharge to coastal waters can be controlled. However, no simple solution exists to deal with the huge number of CSOs in operation, especially in Wales. To manage aesthetic quality it is essential to be able to measure it. Unless aesthetic indicators are identified and aesthetic quality appropriately managed, the effect on the beach users perception of achieving a high quality beach environment through traditional determinands will be severely limited.

6.e.7 Discussion of Results on Seaside Awards for both Surveys 1995 and 1996

The overall concept of beach management can be recognised through the establishment of beach award schemes. Seaside awards are a good idea in principle, providing impetus to actively encourage beach management and provide valuable information to the public. In general the criteria which underpin these schemes are based on safety, management, cleanliness, information and water quality. The most prominent system operating in Europe is the European Blue Flag, introduced in 1987 by the FEEE. In the UK the Blue Flag is co-ordinated by the Tidy Britain Group (TBG), the national independent litter abatement agency. The TBG also own their own beach flag, under the title of Seaside Award, aimed at both resort and rural beaches, requiring bathing waters to meet the less stringent European Bathing Water Directive Mandatory standards. In addition the Marine Conservation Society publish an annual Good Beach Guide, grading British beaches.

Although the aims of the discussed beach award schemes are commendable, their profusion has created much perplexity leading to continued debate over their effectiveness in marketing of beaches. Results of this research conclusively prove that beach users at all three destinations were confused about their purpose, with very few having any accurate level of understanding with respect to their design criteria (Nelson and Williams, 1997). During both surveys at the three beaches approximately only half of the respondents claimed to have heard of seaside award schemes. Just over 20% identified beach flags with indicating either safety or danger and only 15% of respondents ranked attainment of a beach award to be important when asked to compare with other beach attributes such as views and landscape, and ease of access. A higher percentage of the total sample displayed a greater awareness of the Blue Flag compared to any other award and respondents at Cefn Sidan were more knowledgeable about beach awards than beach users at Langland Bay and Whitmore Bay. The poorest water quality was recorded at Whitmore Bay, which also had the lowest response to beach award questions posed in both surveys.

To preserve and improve coastal tourism in Wales, the Wales Tourist Board in conjunction with Welsh Water have set up a relatively new programme, the Green Sea Initiative, designed to improve coastal waters and promote sustainable tourism within the Principality (WTB, 1997b). The intention is to implement high technology ultra-violet light disinfection sewerage systems around the coast, bringing bathing waters up to European Bathing Water Directive Guideline standards (Welsh Water, 1996c). Promotion of the Welsh coast is to be marketed through the achievement of 50 European Blue Flags by the Millenium (Owen *et al.,* 1997).

The Green Sea Initiative and the future of cleaner bathing waters is very promising and will surely benefit coastal tourism in Wales (Owen *et al.,* 1997). However, there are two *caveats*. First is the lack of recognition and knowledge regarding beach award schemes, on which the Green Sea Initiative is reliant and secondly the applicability of the Blue Flag to only resort beaches, of which Wales has few in contrast to its high volume of rural beaches. Results of this study indicate that the volume of beach award schemes is only serving to confuse the beach user at present. If the WTB and Welsh Water maintain faith in the Blue Flag to market the Green Sea Initiative it is proposed an effective and intensive education programme be implemented to create greater awareness of beach awards. Also if investment into the European Blue Flag is to be perpetuated it is recommended that the flag be redesigned to be more representative of the beach environment and pressure applied to the FEEE to reform the Blue Flag to also cater for rural beaches.

6.e.7.1 Proposed Unified Seaside Award System

This research challenges the 'top down' approach in designing seaside awards as being effective. In future more emphasis should be placed on beach user preference, driving beach marketing systems from a 'bottom up' approach. Work on beach rating schemes and beach consumer perception of coastal quality has been conducted by Morgan *et al.* (1993), Williams *et al.* (1993), Morgan and Williams (1995a) and Williams and Nelson, (1997). It is suggested that all existing seaside award schemes are scrapped and replaced by a unified European flag which accommodates both resort and rural beaches. A tiered system is proposed dividing beaches into three groups:

Group A - resort with extensive amenities, such as a funfair
Group B - non-resort with good access, refreshments and toilets
Group C - rural with no facilities provided

Such a flag should be based around the prime categories currently in operation, including safety, management, cleanliness, information and water quality. However, the design must be more sympathetic to public preference, replacing the previous intellectual approach used in developing seaside award schemes. This system should also cater for beaches with varying degrees of water quality, based on the EC Bathing Water Directive (1976a). Two levels are suggested for each group, the EC Mandatory and Guideline standards. This allows for promotion of beaches, without being restricted primarily to water quality, such as the MCS Good Beach Guide (MCS, 1997a). A pleasurable visit to the beach does not necessarily revolve around contact with seawater. Therefore, each group will be sub-divided to accommodate both standards of water quality.

The design of the flag itself should be representative of a beach and each respective Member State should have their own national flag displayed as a component of the beach flag, in one of the corners. To further improve the proposed flag system, a designated agency within each Member State should be assigned to providing additional information at each beach, such as whether it has local conservational interests such as Special Sites of Scientific Interest and interesting walks. In the UK this could be a combined function of the TBG and MCS for example. For any seaside award system to be effective it is imperative that in conjunction with a good design it is coupled with a well marketed education strategy.

Chapter 6(f) Code of Conduct

6.f.1 Introduction to Code of Conduct

The draft Code of Conduct suggested is to provide a practical tool for conducting beach surveys to aid coastal management. Research along the South Wales coastline provided the basis for designing this protocol; the main aim is to form a platform to be developed upon, taking into account local conditions. However, it was not the intention to tackle the regulatory framework through which the coastal zone is managed. Areas not covered in the Code of Conduct include physio-chemical aspects, which need development and inclusion.

The WHO/UNEP (1991) have outlined the need to utilise a multi-disciplinary team in conducting epidemiological-microbiological investigations and also the WHO (1990a) have highlighted the importance of managing the aesthetic quality of recreational waters. The Code of Conduct attempts to address these domains and take these views further, identifying the following key areas:

i. health risk from bathing in marine recreational waters
ii. water quality, investigating the microbiological quality
iii. aesthetic quality indicators
iv. public perception to beach pollution
v. beach marketing tools
vi. beach management

6.f.2 Pre-Survey Design Considerations

- The objectives of the survey should be clearly identified and achievable.

- An audit of resources necessary for the study should be undertaken to ensure sufficient capacity is available to facilitate the research, including logistics, economics, staff, equipment, travel, laboratory capacity and computing power.

- Field work should be carefully planned, defining recreational destinations to be measured and date on which survey will be done.

- The sampling frame must be decided upon and the size of the sample should be sufficient to produce statistically significant results.

- Careful consideration of statistical techniques should be given to study design to validate findings.

- Survey notes should be recorded on all survey days including, environment conditions, tide times, size of tide, air and sea temperatures, winds and visitor load.

- Pilot study should be conducted to trial survey techniques.

6.f.3 Epidemiological-Microbiological Analysis

6.f.3.1 Water Sampling

Sampling Sites and Frequency of Sampling

- The position of all sampling sites should be accurately recorded for consistent measurement.

- Water sampling should reflect temporal, spatial and tidal variations and other local environmental conditions.

- All water samples should be taken as close to the predefined sampling points as possible.

- Water sampling should be conducted at times of highest swimmer density. Research has identified a window period between 11.00am and 3.00pm to represent highest swimmer density. This may vary dependent upon site, and can be verified using a pilot study.

- For large beaches sampling sites should be a maximum of 500m apart.

- No standardised protocol for frequency of water sampling currently exists. The sampling regime stipulated in the EC Bathing Water Directive (CEC, 1976a) is 20 per season, averaging one per week; this programme is believed to be inadequate. Samples should be taken at a maximum of two hour intervals at each site during the time of highest swimmer density. This frequency should be increased, resources permitting.

- To obtain a robust set of results three samples should be taken at each site per time period for replicate sampling.

Procedure

- The NRA water sampling procedure suggested NRA (1991).

- All field observations should be recorded to account for variation in environment conditions, including data and time, state of tide, condition of the sea, wind speed and direction and air and sea temperature.

- All samples should be clearly labelled.

- Sample bottles should of 1.5 litre capacity with a screw top.

- All sample bottles must be sterilised before sampling.

- Water samples should be taken with the sampler stood in knee depth of water.

- A 2m sampling rod should be used to distance the bottle from the sampler when sampling, avoiding exogenic contamination. The bottle clamp must be sterilised with medical wipes and the sampler should wear disposable gloves (WHO/UNEP, 1994).

- Water samples should be taken at a depth of 30cm from the surface of the water, with the mouth facing the current.

- A gap of 20mm should be left at the top of the sample bottle to allow for mixing (HMSO, 1994).

- All efforts should be taken to avoid disturbing the seabed sediment.

Storage

- The samples must be immediately transferred to a thermoisolated box, away from light and transported straight to the laboratory. All sample bottles should be pre-labelled for reference. Analysis of samples should take place between 4-6 hours after sampling (HMSO, 1994).

6.f.3.2 Microbiological Analysis

Bacterial indicator organisms are designed to indicate the presence of sewage and presence of waterborne pathogens, which occur in natural waters. These pathogens exist in large quantities in sewage and present a health hazard when discharged via the sewerage system to recreational waters. Detection of waterborne pathogens is difficult and expensive. Therefore, the function of bacterial indicator organisms is to indicate their presence.

- The two prime indicator organisms stipulated in the proposed reforms to the EC Bathing Water Directive (CEC, 1997) are *E.coli* and faecal streptococci. The WHO/UNEP (1991) also prescribe *E.coli* and faecal streptococci.

- The proposed reforms to the EC Bathing Water Directive (CEC, 1997) have also created provision for future inclusion of bacteriophages as indicators of sewage. As yet no bacteriophage has been selected, but F-specific RNA bacteriophages have been suggested as an appropriate model for enteroviruses in bathing waters. At present no standardised protocol exists to analyse bacteriophages under natural conditions in sewage and receiving waters, and little is know of their densities therein or in human faeces (EC, 1995).

- Quality control programmes should be implemented to evaluate the methods used for all microbiological analysis.

Microbiological Technique

- Membrane Filtration (MF) and Most Probable Number (MPN) are the two main microbiological techniques used for indicator organism enumeration. Both techniques are acceptable under the EC Bathing Water Directive (CEC, 1976a). The MF is a more precise method than MPN and has the advantages of lower cost of operation and greater speed of obtaining results (HMSO, 1994).

- HMSO Report 71 (HMSO, 1994) details standard techniques for MF and MPN bacterial analysis. The WHO (1989b) also sets out a protocol for assessing water quality of recreational waters.

- Geometric means should be used to describe the data, which generally involves the transformation of the data to log10 values. This transformation reduces the likelihood of abnormally high or abnormally low counts in a small number of samples, having undue influence on the overall mean of a large series of observations and also transforms approximately log normal distributions to normal distributions.

Analysis of E.coli

- It is suggested membrane filtration onto lauryl sulphate broth and resuscitated at 300C for 4 hours; after resuscitation incubated at 440C for 14 hours (HMSO,1994). The colony forming units are yellow in colour (HMSO, 1994).

Analysis of faecal streptococcus

- It is suggested membrane filtration onto Slanetz and Bartley agar and resuscitated at 370C; after resuscitation incubated at 440C for 44 hours (HMSO,1994). The colony forming units are pink, red and maroon in colour (HMSO, 1994).

Analysis of bacteriophages

- No standardised protocol exists to analyse phages under natural conditions in sewage and receiving waters and little is know of their densities therein or in human faeces (EC, 1995). Awaiting developments.

6.f.3.3 Epidemiological Design

- Two main research methods most widely used by the scientific community into epidemiological investigations have been adopted by the WHO (1994c):

   i. where the resource budget is extensive the WHO/UNEP (1989b) protocol outlines the Controlled Clinical trial method. This design randomises a sample of beach users into swimmers and non swimmers. Both cohorts are uniform in composition and follow up surveys are conducted to investigate the differential in health risk between exposed and unexposed populations. Illness is confirmed using clinical analysis. The method is disadvantaged by being restricted adults aged over 18 and survey beaches that meet EC Mandatory levels for ethical reasons.

   ii. The WHO/UNEP (1993) have also adopted the Opportunistic (Prospective) Cohort study for local and low-cost applications. This design utilises beach users that self select their activity through their own volition. Therefore, there is no control over swimmers and non-swimmers. Similar to the Controlled Clinical Cohort method a post beach survey interview is conducted to investigation the differential in illness rates between the two cohorts. This method relies on self-reported symptoms, which should be verified by requesting whether the subject has had a visit to the doctor or required any medication (WHO, 1991). Although self-reported symptoms are not as robust as clinical evidence the method does have the advantages of not being restricted to adults and recreational waters that meet EC Mandatory standards.

- Surveys should be conducted over both weekdays and weekends to assess the whole strata of recreational user type.

- To achieve statistical significance surveys should aim to obtain a minimum sample size of 1000 subjects per beach for small scale surveys. For large scale surveys in excess of 2000 subjects per beach should be aimed at.

- The two main mechanisms to contact subjects post beach survey are postal surveys and telephone interviews. The WHO/UNEP (1993) promotes the use of a telephone interview, indicating their effectiveness in achieving a high response rate. Advantages of a telephone interview are the potential to explain any confusion expressed by the respondent and the ability to probe for answers.

- Post beach survey interviews should be conducted between 7-14 days after the initial interview, allowing sufficient time for most waterborne pathogenic micro-organisms to incubate (WHO/UNEP, 1993).

- Information on potential confounding factors should be obtained from the beach surveys including demographic information and non-water related factors. Typical variables include age, sex, socio-economic status, visitor type (local/day tripper/holiday maker), pre-exposure to recreational water prior to beach survey, exposure to recreational water post beach survey and high risk foods eaten, for example shell fish.

- Pooling of bacterial data from different times, states of tides and spatial positions should only be done if the values are not statistically different.

- Statistical techniques for health risk analysis from exposure to recreational waters:

  i.Chi-square (χ^2) analysis may be used to investigate whether there is an association between exposure to recreational waters and elevated symptom rates amongst swimmers.

  ii. Odds ratios (ψ) may be computed using contingency table analysis giving a crude risk estimation of which exposure to recreational waters effects chance of illness.

  iii. The Mantel Haenszel method (ψ_{mh}) may also be computed from contingency table analysis, but provides a summary odds ratio from the stratification of swimmers and non swimmers, controlling for confounding factors.

iv. The WHO/UNEP (1991) recommend the use of Multiple Logistic Regression to evaluate the relationship between exposure to recreational waters and risk of illness. Multiple Logistic Regression is the most powerful technique detailed in the Code of Conduct and has the capacity to control for confounding and interaction effects.

- A self administered questionnaire is advised for the beach surveys facilitating large scale surveys and reducing interviewer bias. For the follow up telephone survey an 'interviewer' questionnaire allows the interviewer to probe for answers.

- All interviewers should receive training.

- Random sampling is preferable. However, with the dynamic movement of people on the beach a systematic sampling programme is a more pragmatic approach.

6.f.4 Aesthetic Quality of Beaches and Beach Management

It is well recognised that water quality is measured in microbiological terms with little consideration given to aesthetic quality of the marine environment. In addition most research avoids consideration of the psychological welfare of the beach user. It is vital that aesthetics and beach user perception to beach pollution be addressed in beach management. To demonstrate the importance of aesthetic quality the WHO (1994a) stated that poor aesthetics has shown to imply poor microbiological-chemical quality and increase the rate of reported symptoms of gastrointestinal from bathing. In addition if beach user perception is not effectively managed there is potential to effect tourism revenue.

6.f.4.1 Beach Pollution

- Aesthetic indicators should be identified, for example oil, tar, plastics, bottles, cans, sewage related debris.

- The WHO (1994a) define aesthetic value as free from:

  i. visible material that will settle to form objectionable deposits
  ii. floating debris, oil, scum and other matter
  iii. substances producing objectionable colour, odour, taste or turbidity
  iv. substances and conditions or combinations thereof in concentrations which produce undesirable aquatic life

- No standardised survey currently exists for measuring aesthetic quality of the beach environment. A protocol has been designed and is being piloted to standardise the measurement of beach aesthetics (Earll and Jowett, 1998). It is suggested that beach managers use this protocol to measure beach pollution.

- The ultimate aim of managing beach litter is to tackle the problem at source by changing the attitude of manufacturers by applying public pressure. This can be achieved locally by pressurising local manufacturers.

- Education campaigns should be designed addressing good disposal practice, for example the 'bag it and bin it' campaign.

- Adequate provision of well maintained disposal receptors should be installed.

- In the short term persistent beach debris can only effectively be dealt with using a mechanical raking system, advised for heavily used beaches. On smaller remote beaches stewardship schemes should be encouraged.

- To reduce visitor input education campaigns should be initiated along with care distribution and maintenance of bins.

- Marine borne litter is a serious long term problem. Conventions have been put in place to effect change with limited results, for example the MARPOL Convention (1973/1978). This is an international problem.

6.f.4.2 Public Perception and Questionnaire Design

- To gauge public perception to beach pollution a semi-structured questionnaire is suggested relating to aesthetic indicators, providing the opportunity to elicit pre-determined information and allowing the subject to express their views.

- Questions should be designed to elicit information that is easily transcribed onto statistical software.

- Questions should be designed to lend themselves to well known statistical techniques.

- Sample sizes should be sufficient to produce statistically significant results. The minimum sample should be 30, but the study should aim at obtaining 100 subjects per beach.

- Questions must be asked in a neutral manner avoiding interview bias. Self-administered questions overcome interviewer bias but can lead to a loss of information.

- All interviewers should receive adequate training such that they comprehensively understand the objectives of the survey.

- On approach to potential subjects the interviewer should introduce themselves explaining the aims of the study.

- The standard approach to gauging public perception to beach pollution is to survey *insitu*. However, researchers have shown that to provide a consistent environment not subject to changing conditions, photographs have proved an appropriate surrogate.

Visual Appearance of Recreational Waters

- Colour and turbidity have shown to have a negative impact on the perception of a beach. Where poor colour and high turbidity are natural, education campaigns should be implemented to reassure the public that these are not necessarily indicative of poor water quality.

- The secchi disc measures transparency which is a function of turbidity and is suggested as a tool for indicating turbidity. The secchi disc is a cheap, simple and easy to use instrument.

6.f.4.3 Beach Management

- A co-ordinated integrated approach to beach management is required involving both horizontal and vertical integration. All stakeholders in management activities should be included including all levels of authority and both public and private sectors
- Effective beach management should include the public/beach user in the planning process.

- Beach inspections for pollution both beach and marine should be frequently conducted and contingency plans in place to deal with pollution incidents.

- All pollution inflows including sewage discharge, estuaries, rivers and agricultural run-off should be identified and monitored.

- All conflicting activities in the coastal zone must be resolved and managed, balancing beach user/tourism demand against sustainable planning and conservation. All coastal zone management techniques should be considered, for example temporal and spatial zoning.

***Beach Provision*:**

- Safety cover dependent upon beach usage.

i. Resort beaches should provide lifeguards. Voluntary surf lifesaving clubs should be encouraged.

ii. Rural beaches which receive small visitor numbers should have a minimum of rescue equipment on display.

- Telephones, especially for use in the event of emergencies.

- First aid facilities, especially on resort beaches.

- Signage relating to natural hazards such as dangerous cliffs.

- Public access to information on health risk, including water quality and beach quality.

Beach Award Schemes

- Beach award schemes encapsulate beach management. However, research suggests that beach users have limited knowledge and a low level of understanding with respect to beach award schemes. It is suggested that limited emphasis be placed on beach awards in the short term

- The European Blue Flag appears to have a higher profile than any other beach award and are being used to market Welsh beaches through the Green Sea Initiative (WTB, 1997). It is advisable that beach managers keep updated on the influence of beach awards in influencing the consumer, especially the European Blue Flag in light of developments in Wales. The major design flaw with the European Blue Flag, which must be noted is that it does not cater for rural beaches, of which Wales has many.

Chapter 7 Conclusion

7.1 Introduction

The coast is a symbolic example of our natural heritage: coastal management is a mantra that is increasingly heard both within the coastal academic and practical scene. Sustainable use of the coastal zone, of which the beach is an integral sub-system, requires careful environmental planning. Researchers have investigated isolated aspects of beach systems, for example health risk analysis from bathing (Cabelli *et al.,* 1982; Lightfoot, 1989; Pike, 1994) and public perception to coastal hazards (Smith *et al.,* 1995b; Phillip *et al.,* 1997). However, few attempts have been made to tackle beach management in an overall holistic context (Williams and Davies, in press).

The research process undertaken in this study has resulted in identification of key variables and their interactions operating upon the beach (Nelson and Williams, 1997). Inherent difficulties arise in not just understanding the complex natural and human dynamisms, but also in comprehending their interaction within the beach system, which may well account for the sparsity of literature in this field. Williams and Davies (in press) defined effective beach management as a response to a specific interaction of cultural influences with the physical environment, with a prime objective of developing a sustainable landscape resource. Grant and Jickells (1995) have also suggested these ideas, identifying the intricate coastal marine ecosystem and the effects of human activities on it.

To bridge the void between the natural and social sciences a multi-disciplinary approach was employed investigating physical beach aspects and the perception of the beach user to coastal pollution and seaside awards schemes (Owen *et al.,* 1997). In addition to conventional microbiological indicators of water quality (CEC, 1997), which were objectively measured in relation to health risk, aesthetic quality of the coastal environment proved to have a more substantial impact upon the consumer. The WHO (1994a) and Owen *et al.,* (1997) also support this view acknowledging the importance of the aesthetic quality of the coastal environment. Philip (1990) has also added weight to the argument by pointing to the increasing necessity of developing aesthetic health indicators.

This study worked towards developing a beach quality indexing system, accommodating the perception of the beach user by measuring the aesthetic quality of the coast in conjunction with standard physio-chemical and microbiological determinands. Water quality indexing (WQI) using conventional parameters is common (House and Ellis, 1987; NRA, 1994b; Minchin *et al.,* 1997), but there is a dearth of literature with respect to coastal landscapes, with aesthetics often being omitted from the WQI systems. Burrows and House (1989) attempted to develop indicators of perceived water quality, but their work was on inland freshwaters. By definition an indexing system aggregates individual indicators or measurements which collectively convey information about quality (Craik and Zube, 1976). However, the multi-dimensional and dynamic nature of the beach and sea, and more specifically their interface made development of an indexing system to describe the beach environment very difficult. The main problem was attempting to combine 'hard' physical data with social data, the formats of which are incompatible. Additionally, aggregating information also leads to loss of information, highlighted by Coughlin (1976) who questioned the ability of an indexing system to paint a full picture of a system. House and Ellis (1987) also acknowledged this weakness, as well commenting that indexing is not totally objective. In this study the aim of designing an acceptable beach quality indexing system was replaced by recognising the less tangible components of beach management and exploring methods of creating a flexible management framework through conceptual modelling.

Before addressing the main research issues it is prudent to make clear that a strategy modification was necessary during the survey work, placing greater emphasis on the water quality/health risk study. At the outset of the project the full implications of conducting an epidemiological-microbiological investigation were not understood. The economic and time resources to achieve a representative sample to run the health risk analysis and the level of statistical analysis required was under estimated. The effect has been to emphasise this aspect of the project over other aspects but not to their detriment.

Survey work carried out during 1995, covering perception to coastal pollution and seaside award schemes, microbiological quality of the water and relationship between bacterial density and illness rates amongst swimmers was conducted at Whitmore Bay. Further perception work developed for the 1996 survey covered an additional two beaches, Langland Bay and Cefn Sidan. Additional microbiological work at Whitmore Bay was done in 1996 along with turbidity measurements at the three beaches. Results obtained at the three identified beaches provide an insight to perception of coastal pollution, but should not be taken as a representative sample of UK beaches. South Wales was used as a case study area to provide a base for further beach management work.

7.2 Conceptual Modelling of Beach Management

The beach environment, as already stated, consists of a complex and dynamic interaction of human and bio-physical processes, which may be the reason why no attempt at modelling the system was apparent in the literature. To achieve successful beach management it is imperative that an holistic view of the beach system is taken by delineating the operative functions. This research has resulted in an attempt to model beach management describing the most pertinent set of variables acting upon the beach environment, recognising their interdependency but also their quasi-autonomous status. Development of Model 1 (Figure 7.1) has responded to the requirement of recognising both the theoretical and practical applications of beach management by modelling the stakeholders, issues and management implications by diagramatically representing the system in three phases:

i. phase 1, (input stage) highlights the stakeholders, main issues and resolutions
ii. phase 2 describes the research process to quantify the main issues
iii. phase 3 (output stage) concludes the findings from the research process by setting objectives to achieve sustainable management planning

The conceptual model is a dynamic function providing a control loop to relay feedback information to the decision makers. This overcomes inherent weaknesses of linear models.

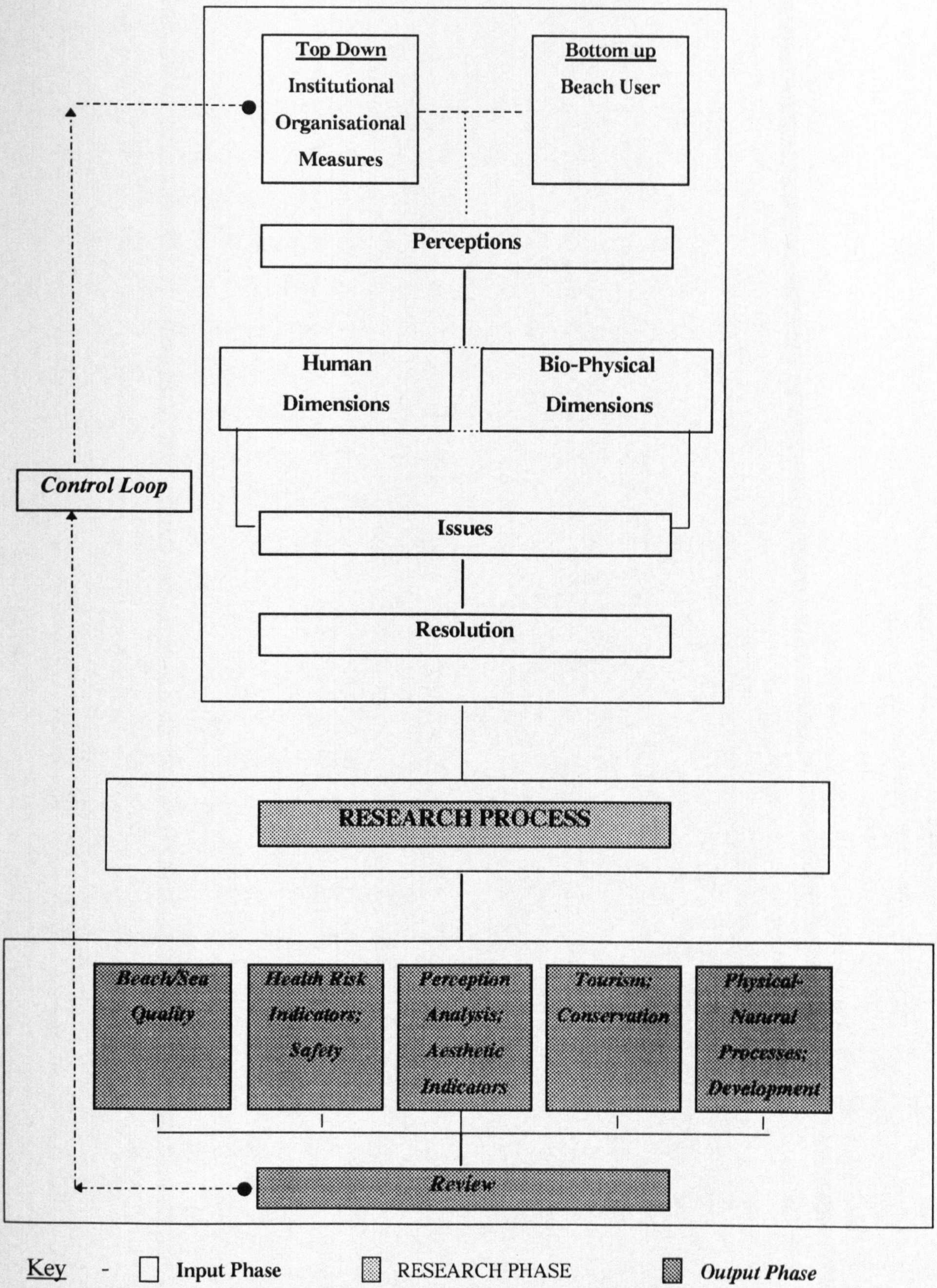

Figure 7.1 Model 1 Conceptual Model of Beach Management

(See Figure 7.2 and Tables 7.1 & 7.2)

7.2.1 Input Phase

Currently UK management frameworks which effect beach management are derived from a 'top down' approach (Figure 7.1), occurring at an intellectual and institutional level, with regulation/legislation being formulated at different levels of hierarchical strata. This thesis highlights the lack of communication between these levels strata, with each component containing individual agendas, which needs to be overcome. In addition the importance of including the beach user in the beach management process is pinpointed, incorporating the perceptions of the consumer from a 'bottom up' perspective into the planning process. It is necessary to re-iterate that the beach is a sub-system of the coastal zone, and any beach management cannot occur without being part of an integrated coastal management (ICM) programme. However, it is not the intention of this thesis to consider ICM, but to recognise the management framework which has a responsibility to beach management.

Model 2 (Figure 7.2) represents the input phase to Model 1. The stakeholders involved in beach management are identified, signifying the necessity for both vertical and horizontal integration and creation of an operational communication link between organisational/institutional level and beach users if successful management is to be achieved. Four levels of management have been classified together with the beach user. It is essential that these components do not operate in isolation, but in a cohesive and integrated mode. The stakeholders are defined below, with specific regard to the UK:

1. Supra-national level - it is important at this top level to:

i. set international directives, notably the EC Bathing Water Directive (CEC, 1976a) and the Urban Waste Water Treatment Directive (CEC, 1991) for protection of health.

ii. design beach award schemes (FEEE, 1997) which encompass beach management, creating a motivational force to improve beaches across Europe.

It is also at this level that protocols for beach management research are designed, for example the WHO and the United Nations Environment Programme (UNEP) for epidemiological-microbiological investigations (WHO, 1989b; WHO/UNEP, 1993).

2. National level - the Department of the Environment (government agency) is responsible for setting regulation to implement EC directives. The Environment Agency (national state governmental agency) have been empowered with the task of monitoring the health of the environment to ensure compliance with EC directives. The TBG, a non-governmental agency are the national litter abatement agency who are also responsible for managing beach awards.

3. Regional level - water authorities (regional agencies) are responsible for ensuring high quality bathing waters and tourist boards (regional development agencies) are responsible for regional economic development.

4. Local level - district councils (local state agencies) operate at ground level responsible for 'hands on' beach management which includes the day to day functioning of a beach, including activities such as beach cleansing and provision of safety cover on beaches.

5. Beach User - primary beach users include recreationalists/swimmers and waterspout enthusiasts. There are a wide range of other miscellaneous groups, which would include for example fishermen, coastal climbers and ornithologists. It is vital that the perceptions of the beach user be included at all levels of the planning process.

Model 2 also defines a set of resultant issues derived from the interaction of human and bio-physical processes, which need resolution. There are numerous management tools for resolving issues including research, delphi technique and discussion/forum groups. Research was the most appropriate technique applicable to resolve the issues highlighted by this study. It was beyond the confines of this thesis to investigate physical and natural processes, such as cliff and beach erosion; conservational issues such as protection of sensitive flora and fauna and coastal development issues which is a component of coastal zone management.

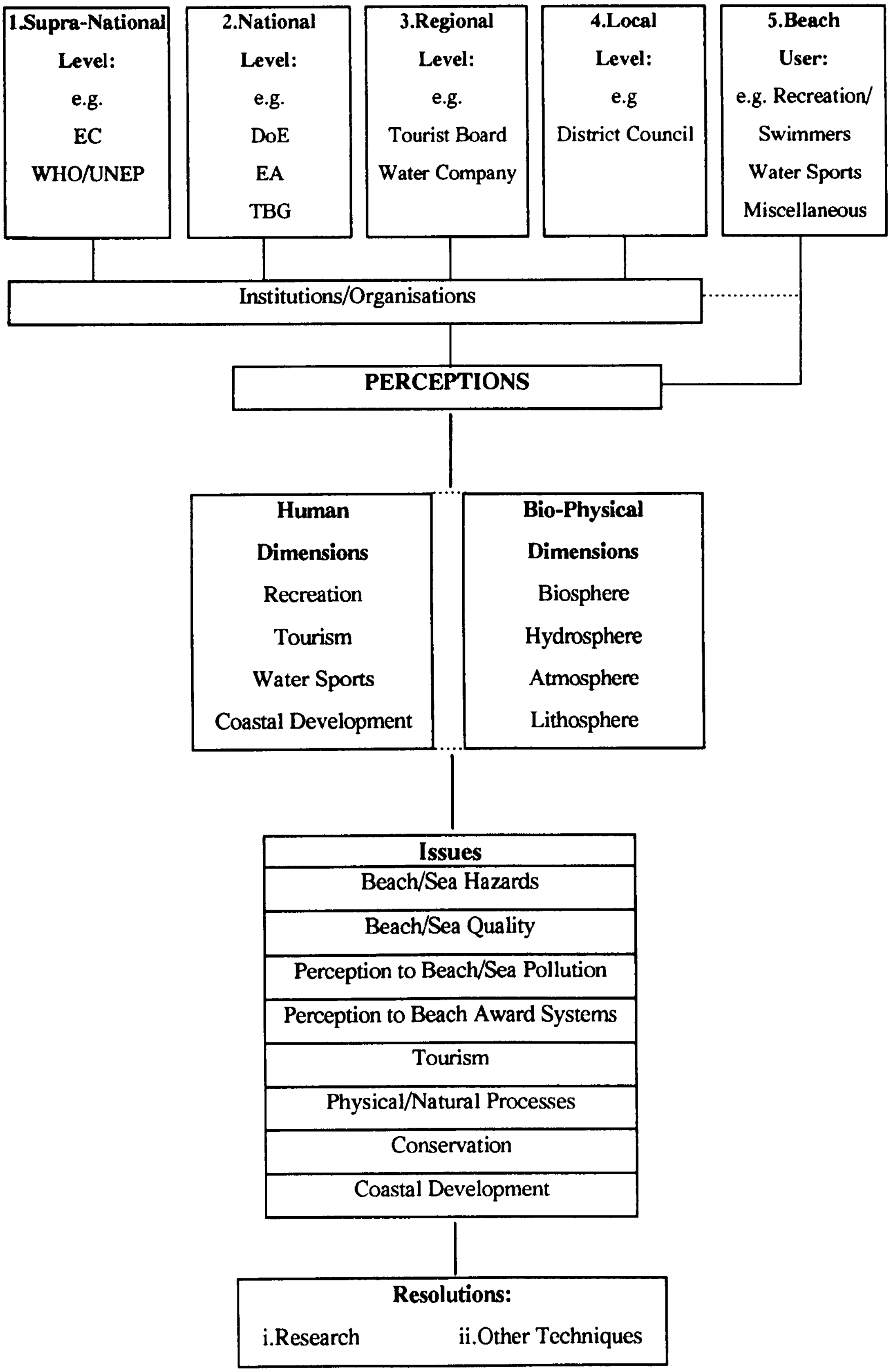

Figure 7.2 Model 2 - Regulation, Dimensions and Issues

7.2.2 Research Phase

The key to successful management is the ability to accurately measure and quantify issues, noted by Earll *et al.*, (1997) who stated 'you cannot manage what you cannot measure'. The beach management issues highlighted in Model 2 are addressed by Table 7.1, which summarises the main findings of this thesis, relating them to other research within the field. A practical 'Code of Conduct' was developed defining an operational working methodology for carrying out beach management research, detailed in Section 6f. The main aim of the code was to establish guidelines for beach managers to investigate quality of beaches and the way in which the public perceive them. The main areas covered include:

i. epidemiology to investigate health effects from bathing
ii. microbiology to examine water quality
iii. aesthetic quality to investigate aesthetic indicators and public perception to beach pollution
iv. beach marketing tools
v. beach management

Information and Education	Information on water quality at beaches is defined in microbiological terminology, which is not understandable in the main to the general public; no information was displayed on beach quality at the beaches investigated.
Management and Planning	The cognisance of the beach user is not currently taken into account in any beach planning or management.
Water Quality Testing	The NRA (1995c) protocol was used for the water sampling process. High counts of E.coli and faecal streptococci were observed at Whitmore Bay during 1995 and 1996. Results of the 1995 water sampling, used for the epidemiological study, yielded average daily geometric counts of *E.coli* exceeding 3300/100ml and average daily geometric counts for faecal streptococci exceeding 426/100ml. These results were significantly higher than the counts obtained by the

	Environment Agency for the same time period and also violate both the current and proposed EC Bathing Water Directives (CEC, 1976a; 1997). High turbidity at Whitmore Bay (average secchi disc readings of 0.92m) was due to its position within the boundaries of the Severn Estuary (Severn Estuary Strategy, 1997). Results showed turbidity to be inversely proportional to distance from the Severn Estuary. Consequently, although the turbidity at Langland Bay (average secchi disc readings of 1.45m) was significantly more turbid than Whitmore Bay, it was more turbid than Cefn Sidan (average secchi disc readings of 1.85m).
Epidemiological Studies	The WHO/UNEP (1993) opportunistic prospective protocol was utilised for the epidemiological-microbiological investigation (this protocol is geared towards low cost local surveillance). The health risk analysis showed a significantly higher incidence of illness amongst swimmers compared to non-swimmers (ψ = 31.37), in agreement with Cabelli (1983), Lightfoot (1989), Alexander and Heaven (1991), Pike (1994) and the WRc (1996a). However, no dose response relationship was observed linking risk of illness to concentrations of bacterial indicators in agreement with Lightfoot (1989) and the WRc (1996a) but at variance with research conducted by Cabelli (1983) and Jones *et al.,* (1993).
Beach Quality	The investigation of litter was formulated around the Norwich Union Coastwatch (Rees and Pond, 1994) method. Transect and quadrat analysis showed the major input of litter to be primarily of visitor source. The major components of the analysis showed plastics and polystyrene to be the most prominent forms of debris on the beach. Hazards associated with the beach include harmful items such as sharp glass and medical waste (Phillip *et al.,* 1997) and pathogenic microbes adsorbed onto beach sediment. The results of this study found no significant proportion of harmful items on the beach, except for a few items of broken glass. No examination of sediment bacteria was conducted in this study.
Perception of	The results of perception towards beach quality were derived from both

Beach Quality	the grid analysis (refer Section 6.c.1) and semi-structured questionnaire analysis (refer Section 6.d.3.3 and 6.e.3). Visual quality of a beach proved to have the most significant effect on beach user perception of quality. In particular sewage related items were found to be the most offensive forms of pollution in agreement with House and Herring (1995). Condoms were found to be singularly the most offensive form of beach debris.
Perception of Water Quality	The results of perception to water quality were derived from semi-structured questionnaire analysis. Discoloured water and high turbidity were found to be major factors in adversely affecting the perception of water quality, similar to findings by David (1971) and Smith *et al.* (1995a). These results matched work by the WHO (1994a) who stated that poor aesthetic quality of waters implies poor water quality to the beach user. Floating objects and oil were also perceived to be significant indicators of poor water quality. The questionnaire analysis also showed beach users' perception of water quality to affect their behaviour. Higher turbidity at the beaches examined led to a greater reluctance to exposure to seawater.
Perception of Beach Award Schemes	There is paucity of literature regarding perception of seaside award schemes in the UK. The results of this study found very limited awareness and understanding of any of the beach award schemes examined. Although the Blue Flag gained the highest recognition of the different systems investigated, beach users in general were unable to identify with the main criteria. A large majority of which failed to recognise the actual flags themselves. These findings were consistent across the three beaches.
Tourism	Tourism and economics related to tourism are of local, regional and national concern. This thesis did not investigate tourism *per se*, except to acknowledge the tourism component of beach awards in marketing of beaches. The WTB and Welsh Water have created the 'Green Sea Initiative' which aims to market Welsh Water's £600m investment into the sewerage system, improving Welsh bathing waters through the attainment of 50 EC Blue Flags by the year 2000 (WTB, 1997b).
Water Safety	Drowning is a significant risk incurred by bathers (Short, 1993).

	Whitmore Bay, Langland Bay and Cefn Sidan all have professional lifeguard provision during the summer months.
Bio-Physical Aspects	Bio-physical aspects such as beach and cliff erosion were not investigated in this study.
Conservation Issues	Conservational issues were not investigated in this study.
Coastal Development	Coastal development is a coastal zone management issue and was not investigated in this study.

Table 7.1 - Research Phase

7.2.3 Output Phase

Table 7.2 defines the output phase of the conceptual model (see Figure 7.1), mapping the way forward for improving beach quality through sustainable planning. Construction of a control loop provides feedback information to the planning and regulation stage (Figure 7.2). This creates a mechanism to adapt to the dynamic system by being able to continually adjust to changing environmental conditions and human activities. Recommendations are made to set environmental quality indicators and suggest management tools for aiding economic development. These include the introduction of a unified beach award system and dissemination of information to the public by means of education programmes.

Information and Education	At present the presentation of water quality results are in scientific terms, such as *E.coli* and faecal streptococci, which the average person finds difficult to understand. Results have shown that beach users are more likely to define water quality in terms of cloudiness and colour. The regulatory authorities have to develop an understandable language to represent the quality of water, and whether it is good or bad, not whether it meets EC Mandatory or Guideline standards. This applies to

	the beach environment in general including the basis on which seaside award schemes are based. Also education programmes should be implemented to include the beach user as part of beach management, encouraging them to be responsible for their activities. The European Charter on Environment and Health also highlight this philosophy, stating that 'every individual has a responsibility to contribute to the protection of the environment, in the interests of his or her own health and the health of others' (WHO, 1989a p.3). Education programmes, for example, should include information of correct disposal of litter on beaches.
Management Planning	This research has identified the need to *include* the perception of the beach user in any future beach management planning operation, including implementation of new legislation/regulation.
Water Quality	Treatment of sewage is the responsibility of the water authorities. The ideal solution to obtaining excellent water quality at beaches would be extermination of pathogens at source (sewage plant), before releasing to coastal waters; a cost benefit analysis would have to be determined. Welsh Water are at the vanguard over treatment of sewage effluent, and have implemented a £600m scheme installing UV treatment plants around Wales (1996a). The result will undoubtedly improve of coastal waters in Wales, and the effectiveness of Welsh Waters initiative should be used as a model to encourage other water authorities to follow suit. The present water sampling requirements stipulated in the EC Bathing Water Directive are inadequate and should be significantly increased and water quality assurance programmes should be introduced to compare results across Member States.
Health Risk Indicators	Until coastal waters are virtually free of sewage, more work is required to identify appropriate indicators of sewage and model health risk. It appears that the relationship between bacterial indicators and health risk from swimming is site specific. Therefore, further work should be carried out at a spectrum of beaches. In the interim period emphasis should be placed on sampling for faecal streptococci which has proved to show a higher correlation with incidence of disease from swimming than any other bacterial indicator (Kay *et al.,* 1994), although not in this

	study. In addition further work on identifying an appropriate bacteriophage to indicate sewage is suggested and research into understanding the pathogenesis of disease (Cartwright, 1993). To standardise epidemiological/microbiological studies the WHO/UNEP (1989b; 1991b) controlled clinical trials should be utilised where budgets are large and the WHO/UNEP (1993) opportunistic prospective protocol for low cost local surveillance research. Beach debris can also cause harm to beach users. The answer to dealing with these items is detailed below in the Beach Quality section.
Aesthetic Indicators	The aesthetic quality of the coastal environment proved to have a significant impact upon the beach user, both on the beach and in the sea. The WHO (1994a) and Williams and Nelson (1997) have identified the need to protect the psychological welfare of the beach user in addition to setting physio-chemical and microbiological determinands to protecting their physical health. Further identification and development of aesthetic indicators is required.
Beach Quality	To manage beach litter it is essential to be able to measure it. The ultimate aim of managing litter is to *tackle it at source*. However, there is no obvious solution to this problem. In the interim period it is suggested that further work be carried out on assessment of beach litter, based on the NALG ABCD model (Earll and Jowett, 1998) to standardise data and build a national picture. Local information provided by the ABCD model could be integrated into the unified seaside award scheme suggested. In the short term it is essential to keep beaches free of litter. The most effective technique is mechanical cleansing of beaches. This should be carried out at resort beaches coupled with adequate provision and maintenance of litter bins (Nelson and Williams, 1997) and education programmes. Stewardship schemes involving litter picks should be encouraged at rural beaches to keep beaches clean whilst protecting the local flora and fauna of the sand habitat.
Perception of Beach Quality	Until beach debris is accurately sourced and long term strategies are implemented to significantly reduce the input of litter onto beaches, the short term solution to alleviating the intrusive impact of litter on the

	perception of the beach user is to regularly cleanse beaches.
Perception of Water Quality	Improvement of the aesthetic quality of the offshore environment is very difficult. Inshore inputs of beach debris through watercourses can be reduced by effecting the source from industry and improving the sewerage system. However, there is no obvious solution to tackling the international transport of litter between countries and resolving the problem of untreated CSOs will have to be a long term strategy. The results showed discoloured water to have a significant impact on the perception of the beach user to coastal pollution and also high turbidity to effect beach behaviour. High turbidity does not necessarily indicate poor water quality. At beaches which are naturally turbid education programmes should be implemented (Smith *et al.,* 1996a).
Beach Award Systems	The lack of knowledge and inaccurate understanding of beach awards questions the applicability of the Blue Flag in representing the success of the Green Sea Initiative and Welsh Water sewerage improvements. The results of this research identify an important public policy dimension, which will need to be addressed if the Green Sea Initiative is to prove successful. This thesis recommends removing all current seaside award schemes and suggests replacing them with a unified seaside award system across Europe. A framework for designing such a scheme is suggested in the main body of the text (refer to Section 6.e.7.1). The proposed system would be to based around the current categories outlined by the EC Blue Flag of safety, management, cleanliness, information and water quality. The main changes in comparison to current seaside award schemes in operation are: 1. Three categories of beach are included for representation by the flag system including resort, non-resort and rural 2. Each category will have two standards of water quality in line with the EC Bathing Water Directive Mandatory and Guideline standards 3. The perceptions of the beach user will be included in design of the final criteria for the scheme 4. Each Member State would be advised to set a regulating agency which would also provide additional valuable information about

	beaches in a guide sheet, for example conservational interests such as Sites of Special and Scientific Interest and interesting walks for each specific site.
Tourism	Tourism and economics are an important part of any beach management plan. There are a variety of agencies involved to some degree in coastal tourism. Tourism strategies occur at varying levels including non governmental organisations, such as the TBG, regional programmes designed by regional economic development agencies, for example the Wales Tourist Board and district councils at local level. This research is not primarily concerned with tourism. However, it is important to note that beach management issues addressed above are important components of tourism, including water quality, beach quality and seaside award schemes. It is also imperative that due consideration is given to the carrying capacity of sensitive beach sites and forward planning occurs to protect sensitive areas and at the same time allow as much freedom to the beach user to fulfil their expectations and enhance their experience to the beach.
Water Safety	Professional lifeguard cover should be provided by district councils at beaches which receive heavy visitor loads. At less frequented beaches local councils should provide adequate rescue equipment, telephone and encourage local voluntary surf lifeguard cover.
Coastal Processes	The coast is exposed to extreme natural elements, which cause significant effects on the beach environment. The shoreline is often the most dynamic part of the earth's surface, being exposed to the atmosphere, the hydrosphere, the lithosphere and the biosphere (Short, 1993). These fields require expert knowledge to investigate their impact on the beach environment, and beyond the confines of this thesis.
Conservation Issues	Although conservational issues were not addressed in this study, it must be noted that conservational issues are an important component of beach management, which requires active managing to achieve sustainability.
Coastal Development	Although coastal development was not addressed in this study, beach management must be considered in any integrated coastal management plan, as coastal development, both industrial and urban can encroach the beach environment.

REVIEW
The review process considers the output of the Conceptual Model (Figure 7.1) and relays the information to the decision makers at institutional/organisational level for evaluation with the aim of monitoring and improving input to the beach management process. This may include and amendment or creation of new legislation/regulation. Again it is important to note the importance of including the beach user as a stakeholder in this process.

Table 7.2 - Recommendations for Sustainable Beach Management

7.3 Summary

The two most significant findings of this study, which are inter-related and frequently omitted from beach management research are:

i. the need to acknowledge the importance of understanding the cognisance of the beach user in evaluating beach and waterscapes, taking into account their experience and expectations and including them in the decision process of beach management.

ii. the requirement to pro-actively develop aesthetic indicators to measure the aesthetic quality of beach environments, which have proved to have more impact on the beach user than physio-chemical and microbiological aspects of beaches.

Coughlin (1976) accurately stated, with regard to water quality, that perception has a reality of its own which is just as valid and perhaps more important in human decision-making than the reality of measurement of physio-chemical and microbiological properties. The WHO (1990) also acknowledged that human aspects should be accounted for in overall an overall management strategy. It is time to implement a system of management based on sound and effective principles which above all include the beach user in the planning process and as part of decision making. Finally, beach management has to reconcile all conflicting interests within beach boundaries to promote sustainable management planning. It is apparent from this investigation that to effectively achieve this aim a multi and inter-disciplinary, co-ordinated and integrated approach in necessary.

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